Optimising carbon storage by land-management

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Abstract

As the UK's largest non-government land-owner, the National Trust is committed to reducing its impact on climate change, recognising the importance of soil organic carbon (SOC), and its need for preservation. To establish if land-management could be optimised to increase carbon storage, 'The Wallington Carbon Footprint Project' was implemented. This study aimed to measure the Wallington Estate's carbon stock, establish what controls SOC, identify carbon under-saturated soils, and make land-management change to increase SOC.

To achieve these objectives a soil sampling campaign and land-use survey were undertaken at Wallington, with further sampling at a verification site in Cambridgeshire. Land-use intervention trials measuring carbon fluxes and SOC change were combined with computer modelling and questionnaires, to assess the impacts of land-use and management change on SOC.

A land carbon stock of 845 Kt (60 Kt within biomass, and 785 Kt within soils) was estimated for Wallington, with the greatest control on SOC identified as grassland land-management. Other controls on SOC were: land-use, soil series, altitude, soil pH and land-use history, indicating that these should be used in all estimates of SOC distribution and stock. A possible link between phosphate fertilisation and SOC accumulation under grassland was identified; however this was not confirmed in a year long field trial. Incorporation of charcoal into soils was identified as a method of carbon sequestration, with a simultaneous reduction in nitrate loss from soil. Surface application to grasslands revealed no detrimental effects on soils, grassland productivity or water quality. Further trials investigated the impacts of arable conversion to short rotation coppice willow, and of peatland afforestation, both indentifying losses of SOC following the land-use change.

Measurement of biomass carbon gains, full life cycle assessment of the each land-use, and the impacts of varying types of biochar are required before firm conclusions regarding land-use change and carbon sequestration can be made.

Declaration and Copyright

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

Signed:

Date:

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ANOVA	Analysis of variance
ANCOVA	Analysis of covariance
C	Carbon
CSS	Countryside survey
CV	Coefficient of variation
DOC	Dissolved organic carbon
GHG	Greenhouse gas
GLM	General linear model
Ha	Hectare
IPCC	Intergovernmental panel on climate change
IRGA	Infra-red-gas-analyser
LOI	Loss on ignition
MLR	Multiple linear regression
POC	Particulate organic carbon
PP	Primary productivity
NER	Net Ecosystem Respiration
NEE	Net Ecosystem Exchange
NPK	Nitrogen, phosphate, potassium compound fertiliser
NSRI	National Soils Resources Institute
NT	National Trust
OC	Organic carbon
OM	Organic matter
PAR	Photosynthetically active radiation ($\mu mol/m^2/hr$)
SOC	Soil organic carbon
%SOC	Soil organic carbon concentration
SOM	Soil organic matter
SRC	Short rotation coppice

Chapter 1

Optimising carbon storage by land-management: An introduction

1.1 Project Rationale: Climate change and the National Trust

The extent to which climate change has occurred since pre-industrial times has gained much greater acceptance over recent years, and the realisation that much of this is the result of human impact is increasing. In recent decades atmospheric carbon levels have risen steeply, culminating in concentrations now 1.3 times pre-industrial levels (Kirby and Potvin, 2007), leading to increasing acceptance of our impact on, and contribution to these emissions. The attitude of the global population is slowly changing, revealed by implementation of the Kyoto Protocol, indicating a global acceptance of the impact that increasing atmospheric carbon concentrations are having on climate change.

This increase in atmospheric carbon is partly due to the burning and use of fossil fuels such as coal and gas (Schwartz and Namri, 2002), and partly due to land-use change decisions made in recent years. The consequences of climate change resulting from increased carbon emissions include potential seasonal and unpredictable temperature change and sea-level rise (Schwartz and Namri, 2002). Global temperatures are predicted to increase by as much as 4.5 °C by 2100 (Updegraff et al., 1998), and the UK is expected to experience much greater extremes of temperature (Hulme et al., 2002). Some of these consequences are already beginning to emerge, leading to the greater acceptance of our impact and contribution to such change, and a realisation that we must act now in an attempt to counteract this change. Developed countries have a commitment as part of the Kyoto protocol to reduce their carbon emissions to approximately 5 % below 1990 levels by 2012 (Worrall et al., 2003), and the UK is committed to reduce its 1990 baseline CO_2

emissions by 8% (Grogan and Matthews, 2001). This translates to a reduction in carbon emissions of 12.7 TgC over the period 2008-2012 (Grogan and Matthews, 2001).

In early 2007 the NT set out to reduce its carbon footprint, realising the importance of its role as a major tourist attraction to lead in reducing the impact we as a population are having on climate change. The NT is a registered charity, conserving buildings, footpaths and the outdoor environment, as well as a major visitor attraction, providing day trip locations and holiday destinations for tourists. Not only are the NT's properties a major visitor attraction, but the organisation is also the largest non-government land-owner in the UK, owning more than 263,000 hectares of land. Much of this land is farmed and is made up of agricultural estates across the UK. Other areas of land owned by the NT are covered in woodland, both natural and plantation forestry (some under the control of The Forestry Commission), and much of the land is open moorland containing some large areas of peat. In addition, several of the NT's estates are home to a number of residents. The nature of the organisation and the combination of carbon emissions from visitor travel, estate and building maintenance, heating and lighting, farm machinery, farm buildings, farm animals, vegetation and soil, makes the NT's total emissions considerable.

In an attempt to reduce these carbon emissions and their impact on climate change, targets have been set by the NT to become more energy efficient, to convert to forms of renewable energy, and to conserve the carbon stored within its land, in both soils and above-ground biomass. To fulfil its aim to combat climate change, the NT is considering many possible methods and attempting to target each of its emission pathways. Projects initially under consideration included implementation of wind turbines, installation of building insulation, encouragement of reduced energy use, sustainable travel and the use of biomass fuels.

The initial aim set by the NT was for the organisation as a whole to become carbon positive, by emitting less carbon to the atmosphere than they accumulate. It was realised that a certain amount of carbon emissions would be inevitable, due to the nature of the organisation, therefore to become carbon positive required focus on enhancing the performance of carbon sinks and creating new carbon sinks, to counteract unavoidable emissions.

1.1.1 The Wallington Carbon Footprint Project

A 3 year long pilot project was implemented in the spring of 2007 to allow the NT to put its aims of carbon emission reduction to the test. The pilot project: "The Wallington

Carbon Footprint Project", was considered as a way to assess any potential problems that may need to be overcome before the aim to make the entire NT land and properties carbon positive can advance. As well as potential problems it was also hoped that new forms of carbon emission reduction and enhanced carbon sequestration strategies could be tested and discovered.

The Wallington Estate in Northumberland was chosen as the location of the pilot project due to its size, diversity of land, extent of visitor travel and number of households. The Wallington Estate is home to 80 households, attracts 180,000 visitors annually and covers a land area of 55 km^2. In terms of reducing carbon emissions and increasing carbon sequestration the focus of the project was not on one specific aspect of the Estate, but would involve the house, gardens, domestic tenants, visitors, volunteers and farmland.

A major aim of the pilot study was to inform policy and practice across the NT as a whole, and to inform visitors and the wider community about climate change and carbon stewardship. The objective of the project as set by the NT was to:

- Carry out a comprehensive carbon audit
- Enhance the performance of all current carbon sinks
- Create new carbon sinks
- Stabilise all carbon stores and sustain stores in favourable condition
- Create new carbon stores
- Reduce the output of all carbon sources – from land, and from all fossil fuel associated emissions

(The National Trust, 2007).

1.1.2 The land based aspect of the project

The land based carbon store is only one aspect of the NT's carbon budget; however it was recognised as playing a major role in whether the organisation could achieve its aim of becoming carbon neutral. The land based aspect of the project is focussed not only on the visible carbon stores seen in the above-ground tree and vegetation biomass, but also in below-ground stores in soils, peat and below-ground vegetation. The remainder of this thesis concentrates on this land-based carbon store and identification of ways in which land-use and land-management can be used, or changed, to reduce emissions from agricultural, moorland and forest landscapes, in order to reduce carbon emissions to the atmosphere and build upon the already large carbon stores in both above-ground vegetation and below-ground vegetation; and in soils. The NT's aim is to ensure that any future land-management change results in reduced emissions of soil and vegetation carbon, and increased soil and vegetation carbon gains.

Alongside their aims to become carbon positive via land-management change, the NT recognise that a large majority of their land is a working environment and that the use of this land provides goods, services and a way of life for many people. Of the entire NT land-holding over 80 % is either farmed or dependent on farming. This fact, along with the recognition that all NT land must continue to support biodiversity, cycle nutrients and water, and produce biomass is something which the NT stress must be taken into account. Any land-use or land-management change for carbon sequestration benefits can not be prioritised at the expense of farming needs, biodiversity and water and environmental quality. The requirement to increase carbon sequestration and reduce carbon emissions via land-use or management change whilst maintaining the functions of the land emphasizes the importance of the land-management aspect of the Wallington project. Alongside building the stores of carbon in their soils, the NT require any future land-uses to reduce their impacts on water, reduce reliance on fossil fuels, enhance and enlarge important wildlife habitats, and reconnect people with the land. They recognise the need to treat water as a precious resource due to the increasing global demand, and also the need to reduce pollution and eutrophication. In relation to reducing fossil fuel use, the NT acknowledges the needs to reduce the use of artificial fertilisers and sprays which are produced using fossil fuel based inputs.

The aims of the NT set out in 1.1.1 must therefore be achieved in combination with meeting the following criteria:

- A reduction in water pollution
- A reduction in fossil fuel use and the use of goods produced via fossil fuel input
- An increase in biodiversity and wildlife habitats
- An increase in access to and use of the land by the wider community

1.1.3 The importance of soil carbon

Increases in atmospheric carbon that the NT are attempting to counteract are largely controlled by the management of soils and the fluxes of carbon into or out of these soils. A loss/gain in soil carbon may result in a gain/loss in atmospheric carbon, and the link between the two means that the management of soil carbon is of vital importance in any attempts to stabilise or reduce current atmospheric carbon levels. Although the aim of this thesis is to identify ways to increase the total land based carbon store including that in the vegetation, the fact that soils store twice as much carbon as vegetation, and two thirds as much as the atmosphere (Smith, 2004), indicates that research into this field is vital and of the greatest importance. Carbon sequestered in SOC is also likely to have a greater permanence than that sequestered in above-ground vegetation, due to the economic returns that sometimes encourage forest logging and above-ground biomass removal (Olsson et al., 2000). The focus of sequestering carbon should therefore be concentrated away from above-ground visible carbon stores which are often considered most important, and towards the below-ground carbon stores which are likely to have a greater permanence. The amount of carbon emitted from soil is very uncertain, and information relating to the spatial distribution of SOC is much sparser than that relating to the spatial distribution of above-ground carbon (Lo Seen et al., 2010). The amount of work undertaken on measuring soil carbon is very limited compared to the measurement of above-ground carbon stores (Walker and Desanker, 2004), therefore highlighting a greater need for more in-depth research into this topic. It is becoming increasingly recognised that atmospheric CO_2 levels are partly governed by the preservation or release of soil carbon (Iqbal et al, 2008), and considering recent concerns over increasing levels of CO_2 in the atmosphere it is essential to ensure that carbon is not being lost from this soil carbon pool.

Although several studies (West et al., 2004, Marland et al., 2004 and Lal et al., 1998, all cited in: Schneider, 2007) have addressed the issues of the non-permanence of soil carbon sequestration, and the fact that soil carbon within mineral soils will eventually reach equilibrium, where they will no longer be able to sequester any additional carbon, it is a vital opportunity that cannot be missed to sequester carbon whilst other methods are sought. Ponce-Hernandez et al (2004) recognise that although soil carbon emission reduction from land use change may not be large enough to stabilise atmospheric CO_2, it can be a stop-gap mechanism whilst other methods are sought.

Increasing the level of OC within soil will also not only sequester carbon from the atmosphere and reduce atmospheric GHG concentrations, but will also increase plant productivity and nutrient and water retention (Anderson – Teixeira et al., 2009), bringing benefits not only in terms of mitigating climate change, but also potential economic and other environmental savings. This is emphasised by Havlin et al (2005), who state that SOC increases soil aggregate stability and hence decreases wind and water erosion. SOC also brings about a reduction in soil bulk density allowing plant roots to penetrate deeper into the soil profile, enabling plant productivity to increase. Soils low in OM are reported to be extremely hard when dry, making them unworkable (Schjonning et al., 2007), having severe consequences for farmers. The potential benefits of increasing soil carbon are therefore many, with not only potential reductions in atmospheric carbon, but also the possibility of meeting the criteria set out in Section 1.1.2 regarding biodiversity, use of the land and water pollution.

1.1.4 The importance of land-management

With agricultural land occupying as much as 50% of the earths land surface (IPCC, 2007) the way we use this land can have major potential impacts. It is widely accepted and reported that SOC stocks differ under different land-uses, and that carbon is released during conversion from grassland or forest to arable land, and accumulated following land-use change in the opposite direction (Howard et al., 1995; Zaehle et al., 2007; Post and Kwon, 2000; Veldkamp, 1994; Guo and Gifford, 2002). In terms of actual land-use change conversion of native land to arable land is thought to result in the greatest carbon emissions, with such a conversion having been reported to cause a 30% loss in SOC (Anderson-Teixeira et al., 2009). Although it is recognised that land-management in addition to land-use change can result in SOC change there is much greater uncertainty regarding the types of management and their consequences. In relation to arable land there are some suggestions that the use of fertilisers can cause a loss of CO_2 to the atmosphere from the soil (Zhang and McGrath, 2004), and in recent years a large amount of focus has been placed on the impacts of tillage on arable SOC stocks. Tillage and SOC disturbance increase SOC decomposition as they expose the OM that would otherwise be protected from microbial attack (Anderson – Teixeira et al., 2009). Conversion from grain to “grain and residue” harvest has been shown to result in a decline in SOC, and some studies have found the greater the amount of residue removal the greater the loss of SOC (Anderson – Teixeira et al., 2009), again emphasising that land-management in addition to land-use change can cause a change in OC. Although the importance of land-management on SOC is acknowledged, its impacts are much less certain than that of land-use change, with particular uncertainties relating to grassland management. It is thus clear that before any land-management and land-use change can advance, more research is needed into the consequences of such action.

1.2 The Study Site: The Wallington Estate

The Wallington Estate is the largest area of contiguous land owned by the National Trust in the UK, covering an area of 55 Km^2. It is located 35 km North of Newcastle Upon Tyne (Figure 1.1).

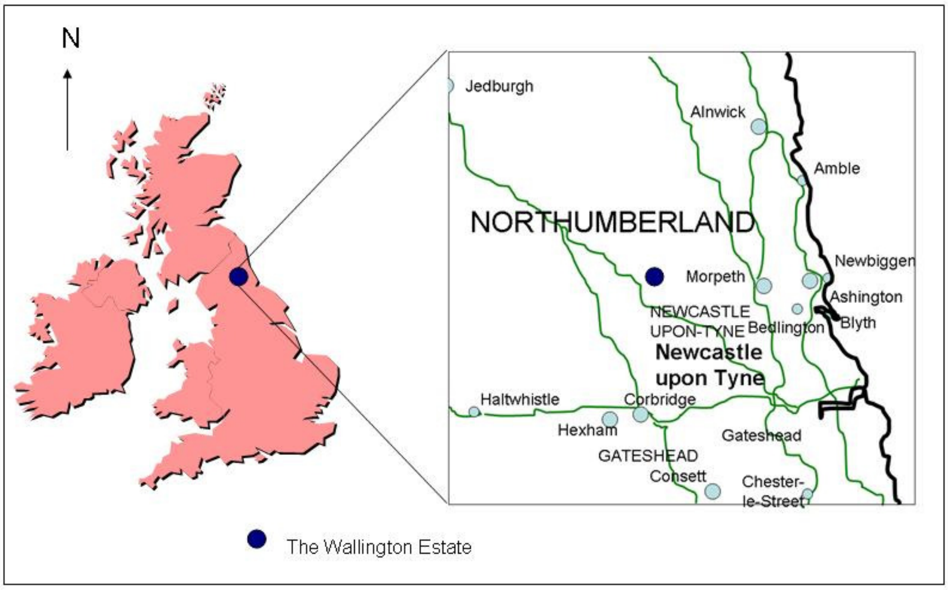

Figure 1.1 The Wallington NT Estate in NE England

Maps showing the extent and variation in land-use, altitude and soil-type across the one Estate have been created in ArcGIS by digitization of paper soil maps (Payton and Palmer, 1989) and records of land-use based on field observations and altitude recorded in the field with a GPS (Figure 1.2). This extent and variation in land-use, altitude and soil type at Wallington make it the perfect location to attempt to identify controls on %SOC typical of at least Northern England.

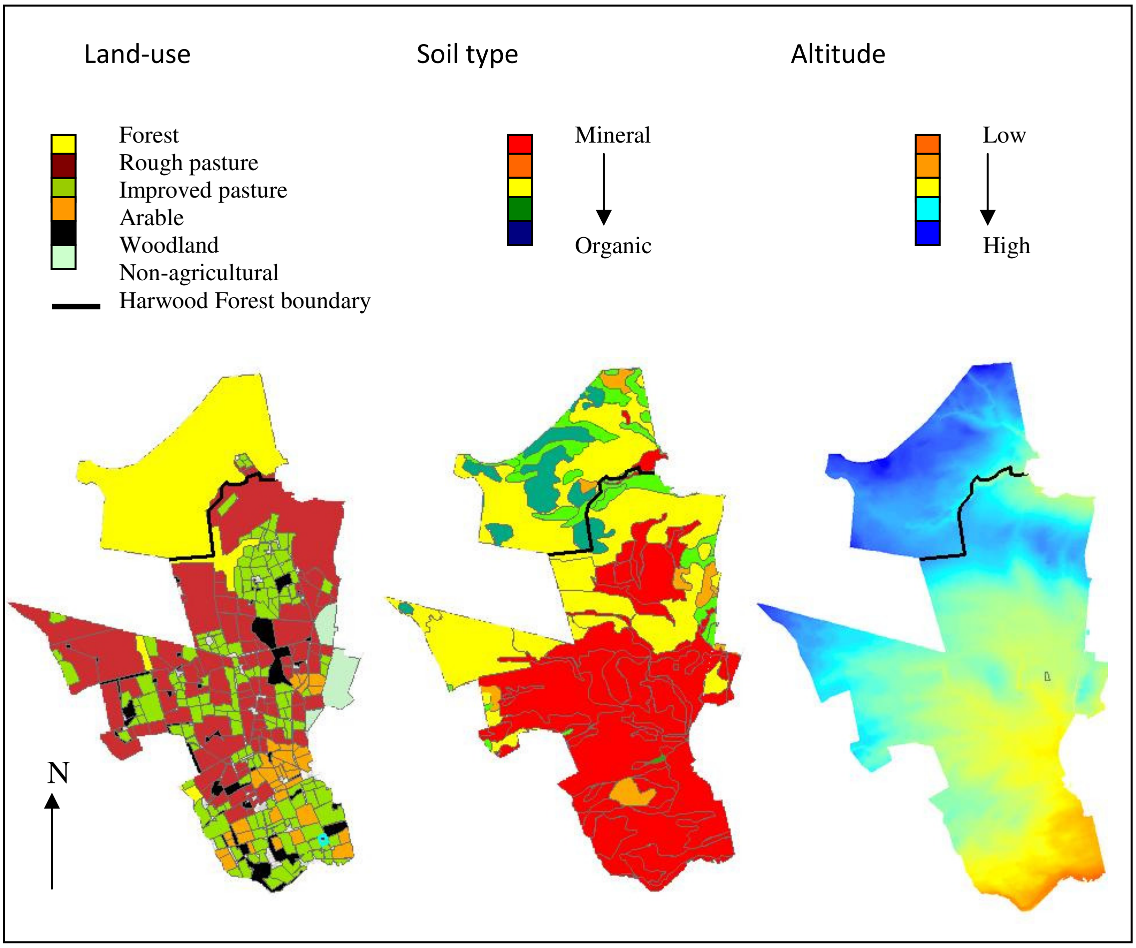

Figure 1.2 The extent and variation in land-use, soil type and altitude on the NT Wallington Estate

The majority of land on the Wallington Estate is leased to agricultural tenancies and a further large component is currently leased to the Forestry Commission (Figure 1.3). Small areas of the Estate are under natural woodland as field margins. Altitude ranges from 100 m in the southern end of the Estate to >350m above sea level in the northern areas under Harwood Forest. The Estate is covered by a range of soil types, including mineral soils, organo-mineral soils (seasonally waterlogged with 15-40 cm thick black surface organic horizons) and organic soils (deep peats with >40 cm thick organic horizons) (Avery, 1980).

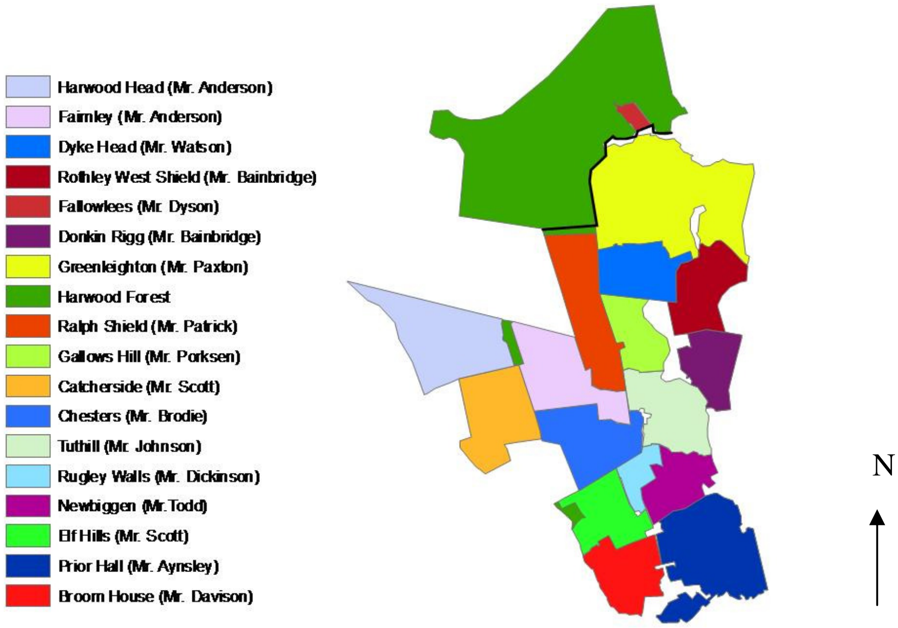

Figure 1.3 The location of farms and land managed by farm tenants on the Wallington Estate

1.2.1 Wallington soil variation

As the largest area of contiguous land in NT ownership it is not surprising that the Estate covers a wide variety of soil types. There are however some particular soil types which are much more prevalent and common than others, as a result of the relatively uniform geological strata in the region and relatively consistent weather conditions across the Estate. The majority of soils are slowly permeable and seasonally waterlogged with a loamy or clayey texture. Towards the southern end of the Estate there are some Brown soils (see: Avery, 1980) which have developed where the ground is free draining, and towards the northern end of the Estate peaty soils and soils with peaty top-soils have been able to develop due to the colder and wetter location.

The variation in soil type across the Estate can be seen by mapping of the individual soil series onto the Estate boundary (Figure 1.4). Comparison of this map with a map of major soil groups created by digitization in ArcGIS (Figure 1.5) shows that although the Estate may have relatively few major soil groups (a single dominant major soil group across the Estate) when identification by soil series is undertaken the variation in soil properties becomes much more apparent. The approximate area of major soil groups on the Estate is: Surface-water-gley: 34.77 Km^2, Brown soils: 7.57 Km^2, Peat soils: 3.24 Km^2, Ground-water-gley soils: 2.79 Km^2, Lithomorphic soils: 0.70 Km^2, Podzols: 0.70 Km^2 and Disturbed/man-made soils: 0.52 Km^2 (see Avery (1980) for a description of each soil type).

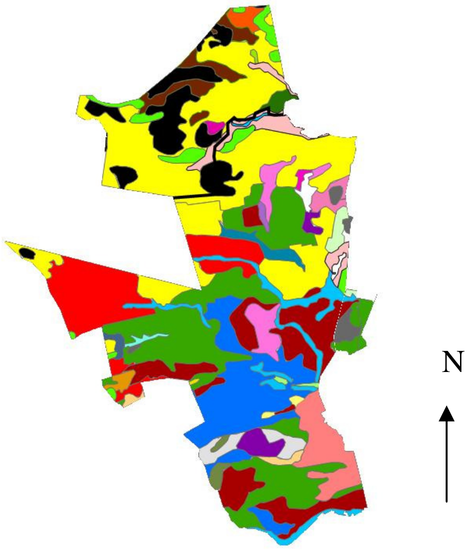

Figure 1.4 The variation in soil series on the Wallington Estate

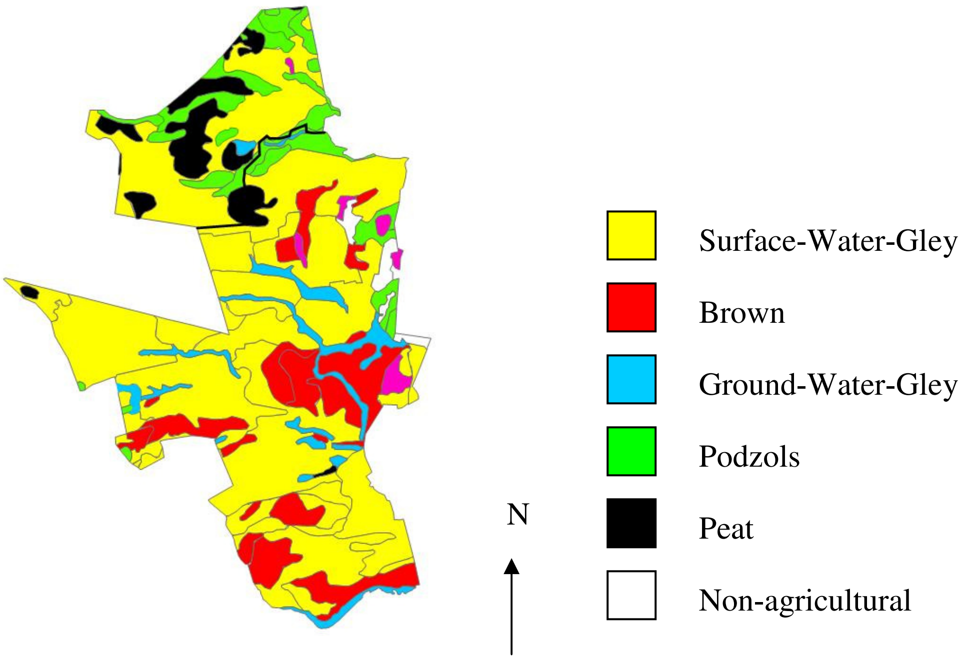

Figure 1.5 The variation in major soil group on the Wallington Estate

1.2.2 Wallington land-use variation and farm locations

The Wallington Estate was initially chosen as the site for the pilot project due to its extensive range of land-uses across the one Estate (Figure 1.2). The Estate is covered by a large area of agricultural land, managed as either arable, improved pasture or rough pasture. The extent and nature of grazing on the pasture land is varied, with both sheep and cattle fields respectively, and further fields containing a combination of both. Some pasture fields are also cut for silage or hay on a regular basis. This agricultural land is currently under the management of 15 tenant farms (Figure 1.3), however prior to 1970 there were a total of 18 tenant farms. The large component of land leased to the Forestry Commission currently operates as a commercial coniferous plantation consisting of a range of conifer species of varying age in a rotation system. The areas of the Estate under natural woodland as field margins also vary in character, but consist largely of mature oak, birch and ash. A small area to the eastern edge of the Estate is under non-agricultural or forestry land-use and was not investigated in this study. The approximate current area of land-use on the Estate is: arable land: 2.9 Km^2, improved pasture: 19.5 Km^2, rough pasture: 13.1 Km^2, woodland: 2.6 Km^2, forest: 12.98 Km^2 and non-agricultural land: 1.56Km^2.

1.3 Aims and objectives

The overall aims of this thesis are to:

- Measure the Estate's current land-based carbon stock
- Establish the controls on the soil carbon stock
- Identify areas/land-uses which are under saturated in soil carbon and have the potential for greater storage
- Identify land-use interventions to increase carbon storage
- Ensure that the results are transferrable to the entire NT Landholding
- Ensure that any land-management change suggestions can be achieved whilst meeting the criteria outlined in Section 1.1.2.

1.4 Review of soil carbon and land-use literature

Research into the influence of land-use and management on soil carbon is becoming more prominent; however the majority of literature is concerned with reducing carbon emissions from arable land and upland peat soils, with a lack of research on the role of pasture management and mineral soils. With arable land, upland peat, lowland pasture and organic and mineral soils all present at Wallington, along with commercial forestry, it is necessary to know the influence on SOC of each management practice.

Measurement of the Wallington Estate's land-based carbon stock will draw upon the work of other researchers who have approached this task to estimate the SOC stocks of the UK (Howard et al., 1995; Smith et al., 2000), as well as other areas of the world. There has however been no such published task attempted for a specific UK organisation or land-holder. This study is different to much of the previous work in both this and other countries, as it aims to assess the combined carbon stock of both below and above-ground land carbon. This study is also an advance on those which have approached the task of calculating SOC using only a combination of secondary data, and aims to assess if more accurate estimates can be gained with physical measurement of SOC stocks at point locations. The use of secondary data and previously produced maps has been a common tool in many researchers' attempts to estimate soil carbon stocks for their region under study. Bernoux et al (2007) used previously published soil maps and soil carbon values to estimate the soil carbon stocks of an area in Brazil. This research is similar to that of Dobermann and Simbahan (2006) who used and compared the results from both use of secondary data tools and the use of primary collected soil sample carbon values. Many other researchers have taken a variety of approaches to calculate SOC stocks at different scales (Meersmans et al., 2008; Yu et al., 2007; Zhang et al., 2004). Although several attempts at SOC stock estimates have been made for many areas worldwide there is no general consensus as to the best approach to take. This study will therefore provide both the NT and other interested parties with a better idea of the best method to use to estimate both SOC and land-based carbon stocks.

Establishing the controls on SOC stocks is an area of major research; however the consensus on what is controlling SOC is very uncertain. Identification of land-use interventions to increase SOC storage has previously involved research into specific land-use change, however research into land-management change and the ability to increase SOC and land carbon stocks by more conservative land-management change is much less studied and more uncertain. The carbon sequestration benefits of converting arable land to grassland or woodland is generally agreed upon globally, therefore, although these are land-management changes that the NT should consider and can implement, the impacts of more uncertain land-management change impacts need to be assessed, and have not been agreed upon in the literature.

In relation to increasing SOC stocks in soils via land-management rather than land-use change, is an area of emerging research concerning the use of biochar. Research into the application and incorporation of biochar (charcoal produced during the pyrolysis of biomass) is a relatively new area of research and there are still many areas of uncertainty regarding its application (Fowles, 2007). Biochar is different to normal charcoal as the term encompasses char produced as a waste product during the pyrolyisis of wood and other farm wastes. The production of biochar is not the main aim, but rather the biochar is produced in the process of energy generation as a by-product. The biochar produced can then be incorporated into land to lock it away from the atmosphere. In relation to grasslands the impacts of fertiliser application and management are very uncertain, and there is very little literature on the role of grazing regime and management practices on carbon sequestration in mineral soil (Ammann et al, 2007). This lack of research is also noted by Allard et al (2007), however they stress its importance due to the huge global extent of grassland. The large extent of pasture not only on the Wallington Estate but covered by the NT across the UK makes research into this aspect of critical importance if the extent of grassland is to remain, but the SOC stock of land to increase. There is some consensus and evidence that SOC stocks are affected by grassland management (Soussana et al., 2004), however the extent and form of these effects needs much further research. Although the area of organic soil/peat on the Wallington Estate is small (Section 1.2.1), research suggests that land-use/land-management change on such soil can have major impacts on SOC emissions/sequestration and that this small area of peat must be preserved. There is a large amount of research into the impacts of peat drainage on soil respiration (Byrne and Milne, 2006; Holden et al., 2007; Lloyd, 2006; Holden, 2005; Dawson et al., 2002) and a general consensus that increased respiration occurs with increased drainage. Many studies have found a decrease in soil respiration with drain blocking (Worrall et al., 2007; Dawson et al., 2002; Worrall et al., 2003), however others have found no change (Holden et al., 2007; Lloyd, 2006). As with mineral soils the impact of grazing on organic soils is uncertain due to a lack of research; however Ward et al (2007) found a decrease in soil respiration upon grazing removal. Although the impacts of re-vegetation on the peatland carbon balance have been investigated there is a lack of research relating to the impacts on soil respiration, with Trinder et al (2008) finding an increase. Although managed burning is not undertaken at Wallington, research has found it to cause increases in soil respiration (Dawson and Smith, 2007; Ward et al., 2007; Rein et al., 2009). The impacts of peatland afforestaion are also uncertain with studies finding either an increase or no change in soil respiration (Byrne and Milne, 2006; Byrne and Farrell, 2005; Hargreaves et al., 2003). There is clearly a large amount of uncertainty relating to the impacts of land-management on upland SOC which need to be resolved before any firm land-management change suggestions can be made for the Wallington Estate, and the NT. It is hoped that this thesis can help to uncover some of the ways in which SOC is affected by grassland management on mineral soils, can add to the literature on land-management impacts on arable land, and can help guide future land-use and land-management change on both upland and lowland organic and mineral soils.

1.5 Thesis outline

In order to meet the aims and objectives outlined in Chapter 1.3 this thesis will take the following format:

- Chapter 2 will deal specifically with the initial aim of the thesis: to measure the current land-based carbon stock of The Wallington Estate, contributing therefore to the overall project aim of carrying out a comprehensive carbon audit of the Estate. This chapter will assess several options by which to measure the total land-carbon stock, and suggest the methods that will provide the most accurate estimates of SOC. The second aim of the thesis, to establish the controls on SOC, is also covered in this chapter, as is the identification of those land-uses that are low in SOC and those which have high SOC stores. Variation in SOC stocks below particular land-uses will help to identify areas with potential for greater SOC storage and suggest possible land-management reasons for these variations, and hence possible changes that could be made to enhance these stocks.
- Chapter 3 will look at establishing the controls on SOC in greater detail, and expand on the findings of chapter 2. It will investigate the possibility of land-management effects on SOC and focus particularly on the variation in SOC beneath rough pasture and arable land. A number of methods will be used to attempt to identify possible land-management interventions that could be made to enhance the stocks in areas of pasture which appear under-saturated in SOC.
- Chapter 4 is a further expansion on the findings of chapter 2 and 3. The focus is on increasing SOC in pasture land - as a land-use which appears understudied and which was found to vary greatly in SOC content in chapter 2. The results of questionnaires and hypotheses developed in chapter 3 are put into practice to establish the affects of pasture fertiliser application on SOC.
- Chapter 5 will look at the ability to predict SOC stocks at another UK location using the factors found to explain SOC stocks on the Wallington Estate. As in chapter 2 other sources of information are tested in an attempt to clarify the most important

information needed to predict the SOC stock and areas of under-saturation for the entire NT landholding.

- Chapter 6 will expand on the aims of chapter 5 and attempt to put the results of the Wallington case study into the wider context and model the impacts of land-use change. The specific focus is on modelling the past impacts of UK land-use change, however the model developed can then be used by NT landholdings across the UK to estimate the impact of their future land-use change decisions on their current SOC stock.
- Chapters 7 and 8 will assess the potential to increase SOC in mineral soils currently in arable land-use. They will look at the carbon flux and environmental consequences/benefits of incorporating lump-wood charcoal into soil, and the impacts of converting arable land to biofuels for fossil fuel substitution and biochar production. Chapter 9 will expand on the research into biochar by assessing the consequences of lump-wood charcoal application to grassland.
- Chapter 10 will focus on high SOC stores and assesses the land-use options to protect and enhance SOC in peat. The impacts of peatland afforestation on both SOC and above-ground biomass carbon stocks will be assessed.
- Chapter 11 draws the work from each chapter of the thesis together and summarises and concludes the results. Conclusions are made in relation to the initial aims of the thesis and the aims of the NT in general. Conclusions are made regarding how land-use and management change can be undertaken to sequester carbon and reduce carbon emissions, and areas of uncertainty which require further work are indicated.

Chapter 2

Controls on Wallington SOC distribution and estimates of land carbon stocks

A reformatted and reduced version of this chapter, relating to the estimation of SOC distribution and its controls has been published in the journal *Geoderma*, and can be found in Appendix 1. I carried out all of the soil sampling, analytical work, data processing and data analysis. I wrote the manuscript in its entirety and then passed it on to the co-author for feedback. This feedback helped to improve the clarity of the manuscript but did not alter the interpretations and conclusions. This paper was co-authored by Fred Worrall, who provided support and guidance throughout, giving critical feedback and help in guiding the direction of the discussion.

All observed data relating to soil sample locations (land-use, soil series, major soil group, altitude, aspect, years in current land-use) can be found in Appendix 2 under the heading 'Wallington soil samples', along with %SOC measured for each sample location.

2.1 Introduction

2.1.1 Chapter objectives

The purpose of this chapter is to contribute to the NT's aim of undertaking a carbon audit of the Wallington Estate, and to measure the controls on, and establish the Estate's land-based carbon stock. The focus of this work is on the below-ground carbon stocks in SOC, as soils store twice as much carbon as vegetation, and two thirds as much as the atmosphere (Smith, 2004), therefore contributing a significant quantity to any region's carbon stocks. Although the focus is on SOC, the above-ground carbon stocks have also been estimated to allow a complete estimate of land carbon stocks to be made. This chapter details the methods used to calculate above-ground carbon stocks and %SOC at locations on the Wallington Estate. It assesses the advantages and disadvantages of various methods used to calculate %SOC, and the information needed in order to make the most accurate estimations. With the focus on SOC the aim was to compare the various options available for calculating the NT's Wallington Estate SOC baseline, and to compare the results of soil samples taken from the field with estimates that would be produced if only secondary data were available. The results of this should suggest the important factors needed to estimate %SOC levels, and identify the information needed to accurately calculate SOC baselines for other NT estates across the UK, as well as suggest important variables which need to be considered in any researcher's attempt to estimate %SOC values and SOC stocks.

As indicated in Chapter 1, recent concerns over climate change and increasing levels of CO_2 in the atmosphere are strengthening the realisation that global warming can be alleviated through a reduction in carbon emissions and increased carbon sequestration. The motivation behind this work is the NT's recognition that any action taken by them as an organisation to reduce their impact on climate change will require a % reduction of their current emissions in respect to their overall carbon stocks. It is therefore vitally important that any above-ground and SOC stock estimates are as accurate as possible in order to correctly quantify the emission reductions required. An accurate estimate of SOC stock and its spatial distribution is also essential as it will highlight areas of high carbon storage which should be preserved and protected, and areas of low carbon storage with the potential for increase.

2.1.2 Controls on, and predictions of, SOC: a review of the literature

The difficulty in estimating SOC stocks is revealed by the variation in global stock estimates, ranging from 1000 to 3000 Gt C (Schwartz and Namri, 2002). This is due to the large spatial variability in SOC (Zhi-Yao et al., 2006) and the use of different databases and scales, meaning that further investigation is needed to establish how best to calculate the most accurate SOC stocks (Meersmans et al., 2008).

Krishnan et al. (2007) recognise that several variables are responsible for differences in %SOC; however they state that many countries and regions do not consider these variables in their SOC stock estimates, and instead base their estimates purely on soil-type, using the average %SOC value for a soil unit. Davidson and Lefebvre (1993) raise the issue of how best to calculate SOC stocks, questioning the use of mean values for soil series versus mean values for major soil group, the implications of using different scale maps, and the advantages/disadvantages of making estimates using land-use rather than soil-type mean %SOC values.

China's SOC stocks have been estimated using the soil survey approach. This involved using mean SOC stocks for a soil-type and multiplying by its area. The stock estimates arrived at varied greatly from 50 to 180 pgC (Yu et al., 2007). Similar uncertainties in estimates for other countries have also been observed e.g. Davidson and Lefebvre (1993) also used the soil survey approach but found issues relating to the scale of map used, with a 13% difference in SOC stock estimates accompanying a change in scale from a 1:250 000 to a 1:20 000 map. Kern (1994, cited in: Guo et al., 2006) used three methods: average value for soil group, average value for soil series, and average value for ecosystem. These provided a range of estimates from 621 to 845 x 10^8 Mg for the USA's SOC stock. Liebens and VanMolle (2003) used the average value for soil-type, and secondly the average value for soil-type/land-use combinations and found differences of up to 7% in SOC stock estimates depending on the methods used. Coomes et al. (2002) also used mean values for soil/land-use combinations and applied these to the areas of those combinations. Stratification of an area into categories such as soil-type, followed by multiplication of point measurements from the stratified areas by the land area of the stratification can result in major inaccuracies. The point measurements may have been taken from a small soil inclusion which has not been mapped due to scale (Tompson and Kolka, 2005) and these soil inclusions could have significantly different carbon contents to the soil series/group which they are then taken to represent.

A better method of predicting a region's or nation's SOC stock needs to be established as it is widely recognised that there are often large coefficients of variation in %SOC within a soil order (Wilding et al., 2001; Davis et al., 2004). If the relationships between %SOC and controlling factors can be better established it will provide a more accurate guide to the reliability and accuracy of current SOC bank estimates. The more accurate that models can be made, the less time and money will need to be spent on extensive sampling and analysis to establish SOC baselines.

Krishnan et al (2007) have identified a range of variables controlling %SOC, including pH, vegetation type, land-cover, temperature, rainfall and soil texture. Tompson and Kolka (2005) are among many authors that have expressed the need to identify the spatial controls on %SOC in order to be able to better estimate SOC stock. They found terrain attributes to be a major control and including this variation in the estimation produced a value 2 times greater than using soil survey data alone. Campbell et al (2008) found large differences in an estimate produced by the soil-survey approach and one produced by including temperature, precipitation and land-use history. Factors found in other studies to control the spatial distribution of %SOC include soil moisture, temperature and texture (Yang et al., 2008), elevation (Powers and Schlesinger, 2002), historical land-use (Schulp and Veldkamp, 2008), precipitation (Dai and Huang, 2006) texture, drainage and slope (Tan et al., 2004), forest management practices and land-use age (Schulp et al., 2008), slope aspect, elevation and terrain attributes (Mueller and Pierce, 2003). Although other research has found management practices to control %SOC levels due to different levels of OM input, grazing intensity and soil disturbance (Frazluebbers and Stuedemann 2008; Huang et al., 2007; Venteris et al., 2004), it has not been common practice to include this variable in estimating an area or regions SOC baseline.

2.1.3 Carbon in above-ground biomass: a review of the literature

In comparison to carbon stored below-ground there is much greater consensus regarding the extent and distribution of carbon stored in above-ground biomass, with forests and woodland widely reported to store the greatest above-ground carbon stocks (Dahl and Anderson, 2007; Gingrich et al., 2007; Tomlinson, 2006; Falloon et al., 2004; Milne and Brown, 1997). There are however differences in the above-ground carbon stocks of woodland and forest depending on the type and age of tree species (Wutzler et al., 2007; Milne and Brown, 1997), with carbon stocks increasing as trees age, then levelling off. In relation to carbon stored in arable crops and grassland there is a general consensus that these are significantly lower than that stored within tree biomass. The extent to which arable land has greater biomass carbon stocks than grassland or grassland has greater carbon stocks than arable land is however more uncertain. Adger and Subak (1996) report higher stocks in arable crops than grassland, as does Tomlinson (2006) and Ordonez et al (2008). Gingrich et al (2007) however report equal carbon stocks in arable and intensive grasslands, and Falloon et al (2004) and Masera et al (2001) report higher stocks in pasture than cultivated land. There is also some variation in the above-ground carbon stocks of pasture land depending on the extent of agricultural management, with Adger and Subak (1996) and Gingrich et al (2007) reporting lower carbon stocks in the biomass of improved pasture than rough pasture, and Tomlinson (2006) reporting lower carbon stocks in pasture than natural grassland.

2.2 Materials and Methods

2.2.1 Study site

Investigation into the controls on %SOC distribution and establishment of the above-ground carbon stocks in this chapter refers specifically to the NT Wallington Estate in Northumberland, north east England (Chapter 1.2). The data relating to controls on %SOC refer only to the results collected from mineral and organo-mineral soils, as it is realised that organic soils behave differently and may not be controlled by the same variables. The data relating to calculation of the total land-based carbon stock however uses SOC stock estimates from all soil types on the Estate.

2.2.2 Estimate of %SOC values using soil samples

As spatial variation in %SOC can be very large (Saby et al., 2008), a high sampling density was required. A total of 618 mineral/organo-mineral soil samples were collected during the period September 2007 to May 2008.

For each sample taken in the field a GPS location was recorded and notes of the altitude, aspect and land-use made. Any relevant notes on landscape position (e.g. topographic decline) were also taken as this is recognised to control %SOC (Dick and Gregorich, 2004). The land-use at each sample point was classified into the following categories: arable, improved temporary pasture, improved permanent pasture, rough pasture, lowland woodland and forestry plantation. Classification was made using the NT's biological survey (Hewins et al., 2001) as a guide, combined with subjective observation in the field and information provided by tenant farmers. It was recognised that any soil samples taken would need to be accurate representations of the area in order to provide reliable results (Cook and Elis, 1987). Before entering the field initial references to ordnance survey, soil maps and NT biological survey maps were made to get an idea of the distribution of potential influencing factors within each field and any areas of particular interest. In fields that appeared highly homogenous (uniform aspect, land-cover, altitude, drainage etc.) a simple random sampling technique was adopted. In fields with a heterogeneous character a more intense sampling rate was used. In large fields a stratified random sampling technique was adopted to break down each field into a number of subpopulations and then a random sample taken from each. Stratification was based on topography, slope aspect and vegetation cover. Samples from areas close to field boundaries were avoided due to the possibilities of compaction from machinery resulting in an unrepresentative sample, as were the corners of fields (which may have been sites for crop and fertiliser storage), gate entrances and other unrepresentative areas. Attempts were made to take samples from every field belonging to each tenant farm; however time limitations mean that some fields were un-sampled. It was however ensured that each combination covering >1% of the estate was sampled.

Measurements of %SOC were made by collecting a sample using either an auger or by digging to a depth of 22 cm. A soil sample was then taken from the 18-22cm layer, giving a value for %SOC at a depth of 20 cm across the estate: 20cm was chosen as it is the depth to which SOC in mineral soils is most likely to be affected by land-use change, (Woomer et al., 2001; Cheng and Kimble, 2001; Kimble et al., 2001) and is the depth used in several similar studies (e.g. Nyssen et al., 2008).

The Wallington Estate boundary was entered into ArcGIS, and the NSRI map of the region (1:50000 scale - Payton and Palmer, 1989) was used to create feature classes for soil series and major soil group respectively. Feature classes for land-use and farm tenancy were also created using a combination of observations made in the field and the Wallington Estate biological survey. The mean %SOC values from the 618 soil samples were then calculated for each soil-series, major soil group, land-use category and farm tenancy respectively. This value was applied to the area of each feature class to which it represented.

2.2.2.1 Analysis of %SOC

All samples were dried overnight at 105 °C and stored. LOI and the Walkley-Black wet oxidation method (De Vos et al., 2007) were used to establish the %SOC in each sample. LOI involved placing the soil sample in a furnace overnight at 500°C to burn off all the OM. The soil sample was weighed both before and after being placed in the furnace, and the weight recorded. %OM was then calculated by subtracting the final weight from the weight of the air-dried sample. This provided a value for %OM and was later calibrated against the %SOC value for the same soil sample as calculated using the Walkley-Black method. The Walkley-Black method involved oxidation of OC within the soil to CO_2 using acidified Potassium Dichromate. Any un-used Potassium Dichromate in the oxidation process was then back titrated with Ammonium Ferrous sulphate and %SOC calculated.

Triplicate or duplicate measurements were made on each individual sample. As only 75 % of the OC within the soil is said to be oxidised by this method (Walkley and Black, 1934) a correction factor of 1.3 was applied to the calculated OC value. The procedure involved placing between 0.1g and 0.5g of air dried soil depending on the predicted %OC (using 0.1g for highly organic soils) into a 500ml conical flask. 10ml of 0.167 M potassium dichromate ($K_2Cr_2O_7$) was then pipetted into each flask and swirled, followed by 20ml of concentrated sulphuric acid (H_2SO_4). Each flask was gently mixed and allowed to stand for 30 minutes. Addition of 10 ml of Orthophosphoric acid (H_3PO_4) was made to each flask to help in the determination of the end point of the titration, and the oxidised samples were diluted to 200 ml with distilled water. Four drops of Diphenylamine indicator were added to each flask and the samples then titrated against 0.5 M Ammonium Ferrous Sulphate ((Fe(NH4)2(SO4)2·6H2O)). The same procedure was undertaken on flasks containing no soil to act as blanks. %SOC was calculated using Equation 2.1 a-c. An example %SOC calculation can be found in Appendix 2 under the heading 'Example %SOC calculation'.

$$N(Fe) = 10.5 / B$$ **Equation 2.1 a**

$$\%EOC = 300 \times N(Fe) \times (B - T) / W$$ **Equation 2.1 b**

$$\%TOC = \%EOC \times 1.3$$ **Equation 2.1 c**

Where:
N (Fe) = Ferrous sulphate strength; B = the titre for the blank; %EOC = easily oxidisable carbon; T = titre for sample; W = weight in mg; %TOC = total organic carbon.

Although %SOC for the large majority of samples was estimated using both methods, time limitations meant that some samples were only analysed by LOI. Accurate estimates of %SOC for these samples were however made using a regression equation from calibration of the two methods. This method of applying a regression equation was also used by Garnett et al (2001). The scatterplot and regression equation produced by calibration of the two methods is shown in Figure 2.1.

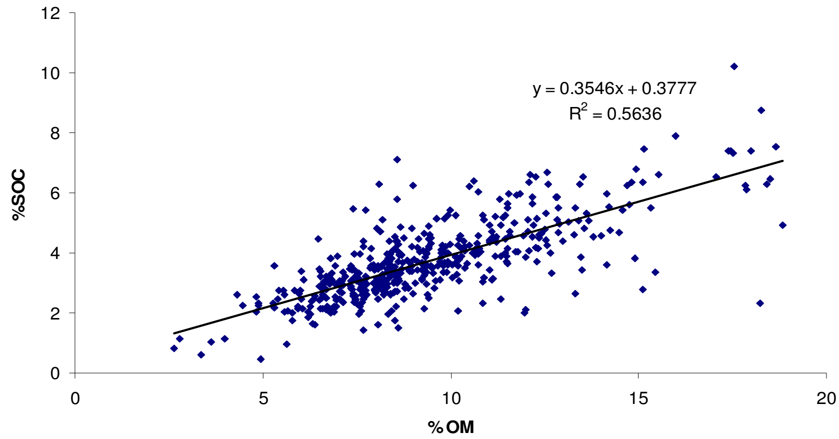

Figure 2.1 Calibration between %OM measured by LOI and %SOC measured by the Walkley-Black method

2.2.2.2 Clay Content and pH

Several studies have found a significant relationship between %SOC and clay content due to chemical protection of microbial decay (Leifeld et al., 2005; Paul et al., 2002; Grigal and Berguson, 1998; Axel Don et al., 2007). As well as providing physical protection, Jones et al. (2004) state that soils with higher clay contents generally have higher %SOC due to greater moisture levels and lower aeration inhibiting oxidation. To establish if clay content is partially controlling %SOC, particle size distribution was measured using the centrifuge method (Tan, 1996). Although a large majority of studies use the Pipette method, time limitations meant the centrifuge method was chosen as it produces results just as accurate as other methods (Tan, 1996). Again due to time limitations all 618 samples could not be analysed for clay content. All samples were entered into a GLM and those under the Soil/land-use combinations covering the largest areas of the estate, with particularly high or low %SOC values for their soil/land-use category were chosen for analysis: in total 160 samples were analysed.

Higher pH results in greater microbial activity (Jones et al., 2004), meaning greater OM mineralisation is expected. Measurement was undertaken to establish if a relationship exists between soil pH and %SOC on all 618 samples. pH was measured using a glass electrode and pH meter following the method of Rowell (1994) and Tan (1996). Although this method of determining soil pH in water will never give an absolute value, comparisons between soil types can be made with confidence (Rowell, 1994).

2.2.2.3 Land-use History (Years in current land-use)

A detailed land-use history was required to assess which soil carbon pools are in equilibrium, and which are adjusting to previous land-use change (Stevens and Van Wesemael, 2008). This was done by interviewing the tenant farmers regarding their land-use from 1980-2008, following the approach used by Nyssen et al. (2008). Limitations at this stage included the fact that some of the tenants are relatively new to the Estate and had to make a best guess of land-use during the earlier period. The results of the land-use survey were entered into ArcGIS and maps of land-use on the Wallington Estate for the years 1980, 1985, 1990, 1995, 2000, 2005 and 2008 were created.

2.2.2.4 Water content

Although the water content of soil could be a significant factor affecting SOC levels, it has not been measured in this study. This was due to the widespread sampling interval spanning September to May, and the realisation that water content would be to some extent influenced by the time of year the sample was taken (Hamer et al., 2008).

2.2.3 Estimate of %SOC using published soil survey data and maps

2.2.3.1 NSRI data

The NSRI soil map of the region (1:50000 scale - Payton and Palmer, 1989) was digitized and the Wallington Estate boundary overlain with each individual soil-series given a feature class. The %SOC contained within the top 20cm of a representative profile for each soil-series was obtained from soil survey publications: this involved referring to soil surveys from across the country to find representative profile descriptions for all soil series present at Wallington. The %SOC contained within the top 20cm of a major soil group was found by calculating the mean value of the soil-series within that soil group. Major soil groups were classified by reference to Payton and Palmer (1989). The mean %SOC for individual soil series and major soil groups identified by reference to soil memoirs can be found in Appendix 2 under the heading 'Wallington soil samples'. The representative soil profiles did include a classification of what land use each soil profile was under at the time of sampling. For a large number of profiles this was permanent grassland, although some profiles were taken under arable, rough grassland and woodland. The land use information was then used to estimate %SOC values for land-uses under which soil series at Wallington occurred, but which were not represented in the NSRI representative profiles. This was done by calculating conversion factors for the limited soil series under which a variety of land-uses were represented in the NSRI database, and applying these conversion factors to all soil series present at Wallington. This was undertaken to investigate if soil-series/land-use combination values would improve estimates of %SOC. The %SOC maps of the estate were then produced by assigning the mean value for that soil series or major soil group to the area of the soil series/major soil group.

2.2.3.2 CSS Data

The CSS database is funded by the Department for Environment, Food and Rural affairs and the Natural Environment Research Council (Countryside Survey data © NERC - Centre for Ecology & Hydrology. All rights reserved). It details information relating to land-use, habitat types and soil data from a random sample of 1km grid squares across Great Britain, and provided 760 point measurements of SOC values from mineral and organo-mineral soils analysed in 1998 and 2000. Major soil group and land-use data was provided for each %SOC measurement, allowing mean values to be calculated for each major soil group, land-use and major soil group/land-use combination present at Wallington. The data from the 2000 CSS data was split into separate land-uses and classified into one of the five land-uses identified at Wallington. The land-use in italics refers to the CSS classification and that in brackets to the new classification: *Crops/weeds* (arable)*; Fertile grassland* (improved pasture); *Infertile grassland/heath/bog/moorland grass/mosaic/tall grassland/herb* (rough pasture); *Lowland wooded* (woodland)*; and Upland wooded* (forestry plantation). Mean values were then assigned to each soil polygon (from the NSRI map), each land-use area (from fieldwork observation and local knowledge) and each soil-type/land-use combination. No data relating to CSS are included in the Appendix due to issues of copyright.

2.2.4 Statistical Analysis

The sampling design conducted within this study could be considered as a three factor experiment with multiple covariates. The three factors are: soil series (and or main soil group); land-use and farm tenancy. All three factors were entered into a GLM as categorical variables using Minitab statistical software. The covariates considered are: altitude, pH, clay content, slope angle and years in current land-use (all continuous variables). This means that the data can be analysed by ANCOVA. Results were considered statistically significant if $p < 0.05$ (95% confidence interval). The results of ANCOVA were post-hoc tested using the Tukey test, and proportion of the original variance explained by factor and covariate was calculated using the method of Howell (1996). Statistical analysis by ANCOVA was chosen as it is a method specifically used on categorical variables, allowing the variability among group means to be compared with the variability between group means. The r^2 values generated by ANCOVA represent the between *group sum of squares* divided by the *total sum of squares,* with a large r^2 value thus indicating that a large fraction of the variation in the independent variable can be explained by the categorical variable/treatment. The r^2 value represents the proportion of the total variation explained by the difference in the means. ANCOVA allows the main effect of the factor/categorical variable to be identified by controlling for the effects of other continuous variables/covariates. This method of analysis removes the effects of variables which modify the relationship between the independent and dependent variables, producing an adjusted mean (an estimate of the true mean if these variables were controlled). To meet the requirement of ANCOVA that all data are normally distributed all %SOC at 20cm depth data were log transformed. Descriptive statistics were used to compare the variability within the different levels of soil or land-use classification.

2.2.5 Estimate of above-ground carbon stocks

Although the main aim of this thesis is to establish the controls on %SOC and investigate the affects of land-use change on %SOC and SOC stock, it is also recognised that a large volume of carbon is stored in vegetation biomass (Schulp et al., 2008). It is therefore important to understand how changes in land-use could affect carbon sequestration as a result of biomass change.

A literature review was undertaken to gain an estimate of biomass carbon stocks. Land-use classifications in the literature were rarely identical to the categories of land-use used in this study; therefore the biomass carbon stock of the most representative land-use from the literature was applied. In studies where grassland was referred to as intensive or extensive these were assumed to be representative of the land-use classes improved and rough pasture respectively. Where biomass carbon stocks were provided for natural grasslands, moors and heathland, these carbon stocks were assumed to be representative of the land-use class *rough pasture* used in this study. As the carbon stocks of tree biomass are also known to vary with tree age (Section 2.1.3), information from the literature was extracted to reveal the carbon stocks for trees of varying age. The age and species of tree present in the land-use *Forest* on the Wallington Estate was calculated using ARC GIS files and information provided by the Forestry Commission (Forestry Commission, © Crown Copyright. All rights reserved 2010). The biomass carbon stocks of deciduous woodland on the Wallington Estate were assumed to equal those quoted in the literature under the land-use woodland, whereas those under the Harwood forest plantation were assumed to equal figures quoted for coniferous forests in the literature. The area of each land-use on the Wallington Estate, and the area of coniferous trees of varying ages were calculated by measuring the area of each feature class created in ARC GIS (see Section 2.2.2). Each land-use was then ascribed a carbon stock value using the area calculated and carbon stock values from the literature.

2.2.6 Estimate of the total Wallington land carbon stock

An estimate of the total land carbon stock required addition of the above-ground carbon stock and the stock of SOC. All methods of estimating SOC referred to so far are in relation to %SOC, however calculating SOC stock requires measurement of %SOC, depth of soil, and soil bulk density (Mestdagh et al., 2004; Tomlinson, 2005). Using this information SOC stock can be calculated (Equation 2.2):

$$Carbon\ stock = \%SOC \times bulk\ density \times depth \times area\ of\ \ soil \qquad \textbf{Equation 2.2}$$

As all soil samples were taken from a depth of 20cm (see Section 2.2.2) this was the depth to which total SOC stock on the Wallington Estate was estimated. As the focus of the work has been on establishing %SOC rather than SOC stock only a limited number of samples were taken for bulk density measurements. These were taken using bulk density tubes of known depth and diameter. The soil within the tube was dried overnight at 105°C and weighed. Bulk density was then calculated by dividing the weight of the dried soil by the volume of the tube. The limited number of bulk density measurements, variability in the results, and difficulty in making such measurements meant that reference was also made to NSRI representative soil profiles and their typical bulk density. In situations where no bulk density measurements were made on a soil series present on the Wallington Estate a typical bulk density for that soil series was ascribed from NSRI representative profiles. Unlike %SOC estimates for un-sampled areas calculated in Section 2.2.1 it was considered best not to calculate total SOC stock by addition of SOC stock mean values for individual land-uses, as the variation in soil series beneath each land-use meant that bulk density would vary significantly within a land-use. Instead it was considered most accurate to calculate total SOC stock by addition of the mean values of SOC stock for each soil series under each land-use. A mean value for both %SOC and bulk density was therefore required for each soil series under each land-use to calculate SOC stock for each soil series/land-use combination. Sufficient bulk density values were however only available for each soil series, therefore bulk density for each land-use under a respective soil series was assumed to be constant. An inter-quartile range on the mean value was also calculated using the inter-quartile range for %SOC for each soil series from samples collected in the field. The area of each land-use/soil series was then calculated from the feature classes created in ARC GIS, and all the information was combined to calculate SOC stock as in Equation 2.2. The surveyed land area of each soil series and land-use can be found in Appendix 2 under the heading 'Wallington land and soil areas'. The total land carbon stock to a depth of 20cm was then calculated by addition of the above-ground carbon stocks and SOC in the top 20cm of soil.

2.3 Results and discussion

When comparing coefficients of variation for individual soil-series/land-use combinations the combinations covering either less than 1% of the estate, or with less than 5 samples were eliminated. During creation of the %SOC maps any combinations of soil/land-use/tenancy which were un-sampled were left blank.

2.3.1 SOC estimate using soil samples

2.3.1.1 Stratification into major soil group

Large coefficients of variation ranging from 16.14% to 48.18% show that there is an extensive amount of variation in %SOC within some major soil groups (Table 2.1), indicating that there is not a strong relationship between major soil group and %SOC, and that other factors are important. The large sample number of 368 for Surface-Water Gley Soils (Avery, 1980) confirms that it is not small sample numbers that are responsible for high coefficients of variation. Although there are statistically significant differences ($p < 0.05$) between some major soil groups, the fact that only 16.18% of the variation in %SOC from samples collected in the field can be explained by the variation in major soil group (Table 2.2) indicates that major soil group alone is not sufficient information to correctly predict any SOC baseline.

MSG (N)	Mean	CV	S Series (N)	Mean	CV	Land-use(N)	Mean	CV	Land use(N)	Mean	CV
Disturbed(8)	5.19	16.14	92(8)	5.19	16.14	IP (1)	4.68		Arable(94)	2.73	24.90
						IT (3)	4.56	24.36	Forest (61)	15.67	28.40
						RP (4)	5.87	10.27	IP(128)	3.70	23.90
Brown (188)	3.32	24.84	Heapey(13)	4.51	37.35	Arable(29)	2.54	17.32	IT(81)	3.10	28.10
			Nercwys(137)	3.23	21.51	IP(47)	3.44	22.49	IP(241)	5.15	39.80
			Waltham(30)	3.14	24.28	IT(32)	3.06	22.38	Wood (8)	4.55	37.19
						RP(80)	3.70	22.84			
GWG (26)	3.84	34.16	Belmont(3)	20.28	12.60	Arable(4)	3.26	30.40			
			Enborne(19)	3.92	29.42	IT(4)	2.26	49.05			
						RP(18)	4.49	26.79			
Lith (4)	8.63	42.04				IP(1)	3.88				
						RP(3)	11.27	37.07			
Pod (20)	14.54	26.25	Cartington(9)	18.70	13.30	Forest(15)	15.41	23.69			
						IP(2)	10.94	29.24			
						RP(2)	9.64	65.40			
SWG (368)	5.05	48.18	Brickfield(125)	3.45	27.36	Arable(61)	2.80	26.30			
			Dunkeswick(33)	3.16	38.88	Forest(46)	15.77	30.09			
			Greyland(53)	3.44	20.69	IP(77)	3.74	20.86			
			Kielder(22)	8.20	35.51	IT(42)	3.14	29.13			
			Ticknall (13)	3.16	23.95	RP(134)	6.19	40.51			
			Wilcocks(122)	9.62	39.06	Wood(8)	4.55	3.00			

Table 2.1 %SOC variation within different soil groups, soil series and soil series/land use categories: Decreasing variation when both soil series and land use class are known. Where: MSG: major soil group, N: no. of samples; CV: coefficient of variation; S Series: soil series; IP: improved permanent pasture; IT: improved temporary pasture; Forest: forestry plantation, RP: rough pasture; wood: woodland; GWG: ground-water-gley soil; Lith: Lithomorphic soil; Pod: podzols; SWG: surface-water-gley soils

SOCmean value Data from:										
NSRI MSG	**NSRI SS**	**CSMSG**	**CS LU**	**Wall MSG**	**Wall SS**	**Wall LU**	**Altitude**	**pH**	**Farm**	**%SOC**
✓										16.85
	✓									48.35
	✓					✓				58
		✓								15.97
			✓							44.75
		✓	✓							51.53
				✓						16.18
					✓					48.69
						✓				45.25
				✓		✓				48.96
					✓	✓				57.72
					✓	✓			✓	64.58
					✓	✓	✓	✓	✓	66.65
					✓	✓	✓	✓		59.27

Table 2.2 The variance in %SOC explained by different data sources and classification methods. Where NSRI: national soils resources institute; MSG: major soil group; SS: soil series; CS: countryside survey; LU: land-use; Wall: Wallington

Table 2.3 shows that this can be expected due to the large range in altitude and land-use beneath the one major soil group. The %SOC map produced by this method is shown in Figure 2.2a.

2.3.1.2 Stratification into soil series

Smaller coefficients of variation ranging from 12.60% to 39.06% indicate less %SOC variability within soil series (see Clayden and Hollis, 1984 for classification) than within major soil groups (Table 2.1) and confirm that soil series is a better predictor of %SOC. Two out of the three soil series within the Brown Soils (Avery, 1980) group have a lower coefficient of variation than for the category Brown Soils, indicating that stratification into soil series is a more accurate method for estimating SOC stock. This is supported by the fact that both soil series within the Ground-Water Gley Soil (Avery, 1980) category have lower coefficients of variation than the major soil group (12.6% and 29.24% compared to 34.16%). Within the Surface-Water Gley major soil group all soil series have lower coefficients of variation compared to the coefficient of variation for major soil group (48.18%).

There is a significant improvement from 16.18% to 48.69% in the ability to explain %SOC variability using the mean value for soil series rather than major soil group (Table 2.2). Statistically significant differences between several soil series further indicate that soil series is having some degree of control on %SOC levels. The %SOC map produced by this method is shown in Figure 2.2b. Reference to Table 2.3 again indicates how this can be expected due to a smaller range in altitude beneath the one soil series than the one major soil group.

Stratification	%SOC predicted	Area (Km^2)	Altitude range (m)	No.of soil series	No.of land-uses	No.of farms
Major soil group	**16.18**	**34.86**	**188**	**7**	**4**	**17**
Land-use	**45.25**	**11.92**	**167**	**22**	**1**	**16**
Soil series	**48.69**	**8.88**	**122**	**1**	**4**	**12**
Farm	**55.46**	**2.47**	**35**	**9**	**3**	**1**
Soil series/land-use	**57.27**	**4.25**	**124**	**1**	**1**	**10**
Soil series/land-use/altitude	**59.27**	**1.41**	**15**	**1**	**1**	**7**
Soil series/land-use/altitude/farm	**66.65**	**0.42**	**15**	**1**	**1**	**1**

Table 2.3 Different levels of stratification, their areal coverage and ranges in other controls on %SOC.

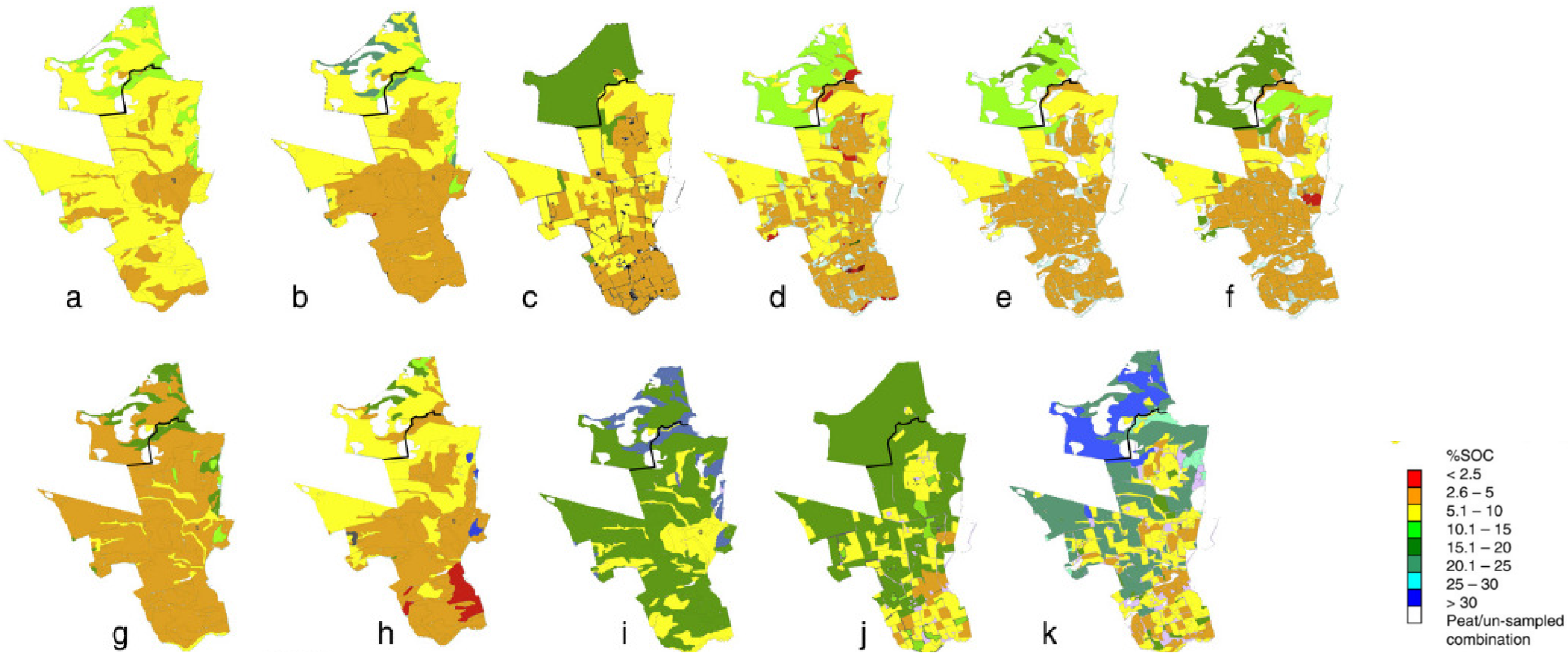

Figure 2.2 %SOC distribution estimated using mean values from: a.) Fieldwork major soil group; b.) Fieldwork soil series; c.) Fieldwork land-use; d.) Fieldwork major soil group/land-use; e.) Fieldwork soil series/land-use; f.) Fieldwork soil series/land-use/farm tenancy; g.) NSRI major soil group; h.) NSRI soil series; i.) Countryside Survey major soil group j.) Countryside survey land-use; k.) Countryside survey major soil group/land-use

2.3.1.3 Stratification into land-use

Coefficients of variation ranging from 23.97% to 39.89% indicate less variability within land-use categories than within major soil groups (Table 2.1, columns 10-12 compared to 1-3), however the lowest coefficient of variation of 23.97% compared to the lowest coefficient of variation for soil series of 12.06% suggests that some soil series have less variation than some land-use classes. This indicates that generally soil series is a better predictor of %SOC than land-use, if this is the only information available. This is confirmed by the lower r^2 value of 45.25% indicating that only 45.25% of the variability in %SOC can be explained by the variation in mean land-use %SOC, therefore less chance of correctly predicting %SOC at a specific location if the estimate is based purely on land-use as opposed to soil series. There are however statistically significant differences between arable and forestry; rough permanent pasture and improved permanent pasture; forestry and all land-uses; improved permanent pasture and rough permanent pasture; improved temporary pasture and rough permanent pasture, suggesting that land-use is having some influence on %SOC. The %SOC map produced by this method is shown in Figure 2.2c. Reference to Table 2.3 shows how the area of the estate covered by an individual land-use is large, therefore covering a large range in altitude and soil series, which will again be responsible for the variation in %SOC beneath a particular land-use.

2.3.1.4 Stratification into farm tenancy

The range in coefficients of variation from 15.28% to 40.91% indicate that some farm tenancies have much less variation in %SOC than others, most likely due to some having various land-uses and soil types, compared to others with one dominant land-use and soil type. An r^2 value of 55.46% suggests that farm tenancy is a better predictor of %SOC than soil series or land-use if this is the only information available on which to estimate %SOC. The majority of tenancies show no significant differences, however there are three farms with significantly higher %SOC values, therefore, although estimation of SOC stocks based on stratification into farm tenancy will produce an estimate more accurate than soil group, soil series and land-use stratifications respectively, this is most likely the result of inconsistencies in the other variables affecting %SOC between farms. Single variant analysis can not establish whether farm management practices are responsible for %SOC variation due to differences in soil series, land use, altitude and other variables between farms.

2.3.1.5 Stratification into major soil group/land-use

The coefficients of variation for all land-uses within the major soil group Brown Soils are lower than the coefficient of variation for just Brown Soils (Table 2.1, columns 7-9 compared to 1-3). Within the major soil group Ground-Water Gley soils, the land-use categories arable and rough pasture have lower coefficients of variation than for the Ground-Water Gley Soil category. Rough pasture within the Lithomorphic Soil (Avery, 1980) category has a lower coefficient of variation than the category Lithomorphic Soils, and all land-use categories within the major soil group Surface-Water Gley have lower coefficients of variation than the coefficient of variation for Surface-Water Gley. This confirms that stratification into major soil group/land-use category would achieve a more accurate estimate of %SOC compared to stratification using major soil group alone. This is also confirmed by the large increase in r^2 from 16.18% to 48.96%. Stratification of the area into Major soil group/land-use categories would also provide a more accurate estimate than stratification into land-use (r^2 = 45.25%) and soil series (r^2=48.69%). The %SOC map produced by this method is shown in Figure 2.2d.

2.3.1.6 Stratification into soil series/land-use

As mentioned earlier, access difficulties and remote areas of small soil-series inclusions meant that a mean value for each soil-series/land-use combination at Wallington has not been measured. The result of this is that the predictive value of using the mean values to estimate %SOC values for these soil/land-use combinations can not be assessed. These areas however tend to cover less than 1% of the estate and therefore inaccuracies in calculating total %SOC levels as a result of this are small.

Within the soil series Breamish, 4 out of 5 of the land-use categories have lower coefficients of variation than Breamish; all land-uses within Dunkeswick have lower coefficients of variation than Dunkeswick, as is the case with land-uses within Greyland, Nercwys and Wilcocks soil series. The large increase in r^2 to 57.72% indicates that soil series/land-use stratification is the most accurate method of predicting SOC stocks if you only have information relating to soil type and land use. The %SOC map produced by this method is shown in Figure 2.2e.

2.3.1.7 Stratification into soil series/land-use/farm tenancy combinations

If however you also know which farm tenancy the land-use and soil series is located under, the probability of correctly predicting %SOC will be improved from 57.72% to 64.58%. The coefficients of variation for all tenancies within the category Brickfield/arable are lower than the coefficients of variation for soil series stratification into Brickfield, and land-use stratification into arable. The coefficients of variation for both Newbiggen and Prior Hall within the category Brickfield/arable are lower than the coefficients of variation for stratification based purely on tenancy. The same is true of many other soil series/land-use/tenancy stratifications. The %SOC map produced by this method is shown in Figure 2.2f.

Although there is a statistically significant difference between many of the farm tenancies under the same soil series and land-use, the possibility that this could be the result of other potential %SOC controlling factors including altitude, pH and clay content must be investigated.

Regression analysis of %SOC against altitude reveals that 41.5% of the variance in %SOC can be explained by altitude. Again however, single variance analysis at such a complex site is insufficient to establish the factors controlling %SOC. Soil series and land-use as well as tenancy are all governed to some extent by altitude. The r^2 value of 41.5% does however reveal that having information only relating to altitude would produce a SOC estimate of greater accuracy than stratification into major soil group alone. Regression analysis of %SOC against pH reveals that 32.8% of the variance in %SOC can be explained by pH: the same issues relating to single variance analysis again however exist. Although it is unlikely that you would have information relating to soil pH and not altitude, land-use, soil series or farm tenancy, if this was the case, you would be able to achieve a more accurate estimate of SOC using pH as a predictor rather than major soil group alone.

Other factors which were thought to be possible controls on %SOC including land-use history (years in current land-use), slope aspect, slope angle and clay content were also included in the model but did not have a statistically significant affect on %SOC ($p>0.05$). The results of a land-use history survey undertaken with the Wallington estate farmers are presented in Figure 2.3 revealing very little change over the last 25 years. It may therefore be possible that land-use history is an important control on %SOC, but that with a lack of any major land-use change on the Wallington estate the impacts of this variable can not be fully investigated in this study.

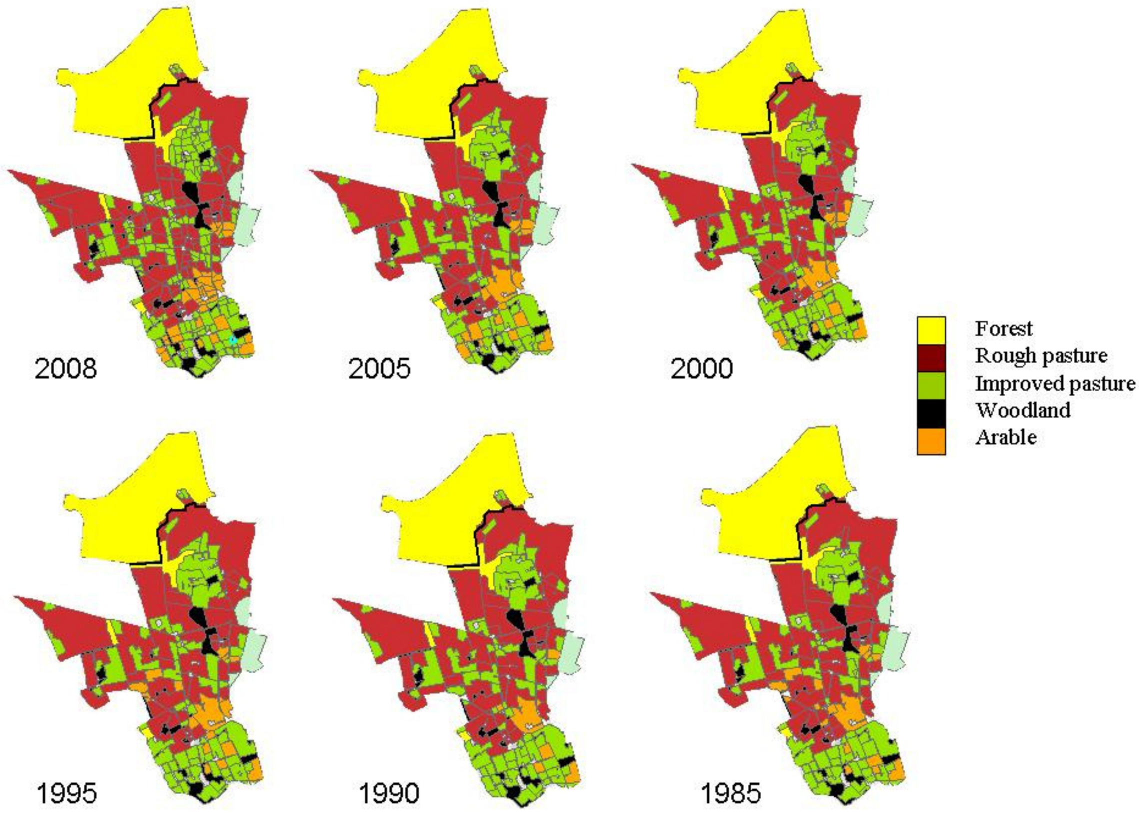

Figure 2.3: A lack of land-use change on the Wallington estate

2.3.1.8 Stratification into Soil series/land-use/farm tenancy/altitude/pH

Inclusion of altitude and pH in the model increased the r^2 value from 64.58% to 66.65% and both factors were identified as having a statistically significant affect on %SOC (Table 2.2).

To assess the impact of classification into farm tenancy on %SOC estimates, farm tenancy was removed from the model (leaving soil series, land-use, altitude and pH), and the statistically significant differences between land-uses were compared to the statistically significant differences between land-uses in the model stratified by farm tenancy (soil series, land-use, farm tenancy, altitude and pH). At a specific altitude, land use, soil series and pH, when tenancy is kept constant there is no longer a difference between arable and improved permanent pasture. This suggests that without the inclusion of tenancy there was a larger spread in the %SOC values found under these categories. With tenancy included there is now a difference between arable and improved temporary pasture suggesting that there was previously a larger spread in the values for these categories and these have been reduced with stratification into farm tenancy. There is no longer a difference between forestry and rough pasture suggesting that the spread of %SOC values for rough pasture are greater across the entire Wallington estate than they are within tenancies, indicating that different levels of management practice within rough pasture are causing differences in %SOC. There is now a difference between improved permanent pasture and rough pasture suggesting that the coefficients of variation for these categories have become more constrained. This again suggests that farm management practices within these categories are controlling %SOC levels. There is no longer a difference between improved temporary pasture and rough pasture. This again suggests a reduction in variation within land-use classes when stratified by farm tenancy.

The role of farm management practices on %SOC is emphasised when the magnitude of the effect of each variable is analysed. When altitude, soil series, land-use, farm tenancy and pH are constant, a change in any of these variables has a statistically significant affect on %SOC. The proportion of original variance explained by each variable was calculated as explained in Section 2.2.4, and is shown in Table 2.4.

Variable	Magnitude of effect (%)
pH	5.14
Altitude	13.47
Soil series	9.17
Land use	19.67
Farm tenancy	35.29

Table 2.4: Controls on %SOC: the greater impact of farm tenancy compared to land-use and soil series: an indication of land-management effects.

These results indicate that farm tenancy has a greater influence on %SOC than both land-use and soil series. The generally good model fit is shown in Figure 2.4; however the prediction of 66.65% of %SOC variation with these variables included means that 33.35% of the variation is still unexplained.

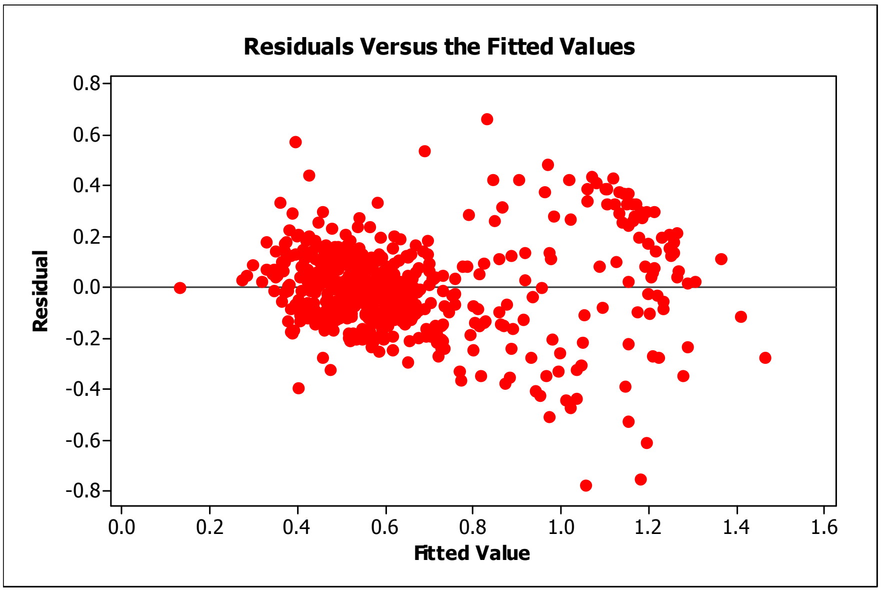

Figure 2.4 Modelled values of SOC versus residuals: using soil series, land-use, altitude, pH and farm tenancy as inputs.

2.3.2 SOC estimate using published soil survey data

2.3.2.1 NSRI data

Regression analysis of the 618 samples versus the mean values for major soil group from the NSRI map series indicates that only 16.8% of the variation in %SOC can be explained by the variation in soil group. The %SOC map produced by this method is shown in Figure 2.2g. Using mean values for soil series from the NSRI map series would produce a significantly better estimate, explaining 48.35% of the variation in %SOC values. The %SOC map produced by this method is shown in Figure 2.2h. However, the r^2 value of 48.35% shows that more than 50% of the observed variation is unexplained, and that other variables must be included. Using mean values for soil series/land-use combinations calculated from a limited NSRI database showed that variations in soil series/land-use combinations could explain 58% of the variation in %SOC, indicating the major importance of land-use on %SOC values.

2.3.2.2 CSS Data

Regression analysis of the 618 %SOC values against the mean values for major soil group from the Countryside Survey database (calculated as explained in Section 2.2.3.2) indicates that only 15.97% of the variation in %SOC could be explained by variation in these values. The %SOC map produced by this method is shown in Figure 2.2i. Using mean values for land-use would produce a significantly better estimate, with 44.75% of the variance in %SOC values explained by the variance in land-use values. The predictive value is increased further still when mean values for major soil group/land-use combinations are applied. Applying these values explains 51.53% of the variation in the measured %SOC data. The %SOC maps produced by these methods are shown in Figures 2.2j and 2.2k.

These results suggest that the CSS database is the more accurate of the two methods for calculating SOC baselines if no local soil sampling and fieldwork is available, and only raw data provided by the two sources is used, however the highest r^2 value of 51.53% implies that other variables are controlling %SOC levels and should be included to achieve greater accuracy. Although the CSS soil/land-use combinations are explaining 51.5% of the variation in %SOC, the green and blue colours in Figures 2.2i, j and k show that the CSS is predicting values that are systematically too high for the more organic rich soils and areas of rough pasture. This is very important when calculating SOC stocks and although the majority of this study refers only to %SOC values rather than SOC stocks, a comparison of SOC stocks at this point emphasises this point. NSRI data for soil series would predict a carbon stock for the top 20cm of soil on the estate of 556.13Kt C, CSS data for major soil group/land-use combinations would produce a carbon stock value of 1188.43Kt C and fieldwork values for soil series/land-use combinations a carbon stock value of 785.24Kt C.

The accuracy of predicting %SOC values can be increased using published data if NSRI data is manipulated and soil series %SOC values are converted to take account of land-use.

2.3.3 Above-ground biomass carbon stocks

A review of the literature revealed mean biomass C stocks of 3 t C/ha for arable crops, 5.2 t C/ha for improved pasture, 2.78 t C/ha for rough pasture, 53.33 t C/ha for woodland and a range from 4.8 – 69.1 t C/ha for forests depending on the age of tree. The spatial distribution of above-ground carbon stocks on the Wallington Estate is displayed in Figure 2.5. Comparison with the land-use distribution map (Figure 1.2) reveals the greatest stocks under areas of Harwood forest and the lowest stocks under areas of rough pasture. All areas of Harwood forest are not however covered by biomass high in carbon stocks as a result of the age of tree species and a number of recently clear-felled areas. The biomass carbon stocks of some areas of Harwood forest are therefore as low as 0 t C/ha. Although the lowest biomass carbon stocks are found under rough pasture, the variation in biomass carbon between rough pasture, arable land and improved pasture is not great. The biomass carbon stock under arable land on the Wallington Estate is currently 897 t C, under improved pasture is 6214 t C/ha, under rough pasture is 6038 t C/ha, under woodland is 13322 t C/ha and under Harwood forest is 33810 t C. This totals a biomass carbon stock for the Wallington Estate of 60.29 Kt C.

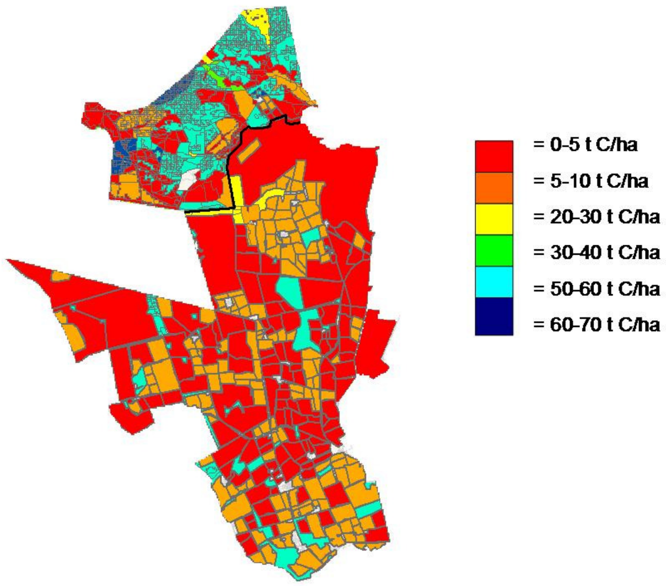

Figure 2.5 The Wallington Estate above-ground biomass carbon stocks

2.3.4 Total land carbon stocks on the Wallington Estate

The total SOC stock on the Wallington Estate contained within the top 20 cm was calculated to be 785.24 Kt C +/- 312.83. When combined with the biomass carbon stocks this totals a land carbon stock for the Wallington estate of 845.53 Kt C. Comparison of the biomass and SOC stocks reveals a much greater carbon stock within the top 20cm of soil than within the biomass.

2.4 Discussion

This study highlights the issues of scale involved in calculating SOC baseline inventories. Examination of Table 2.3 indicates why applying the mean %SOC value for a particular major soil group gathered from an area as large as 55Km^2 will limit the accuracy of the prediction, due to the large range in other possible controls on %SOC beneath that one land-use. The same is true when applying mean values taken from a particular land-use or soil series covering such a large area. Applying mean %SOC values taken from beneath one particular farm tenancy could possibly increase the accuracy of the estimate due to the farm covering a much smaller scale than a particular land-use, major soil group or soil series (Table 2.3), therefore decreasing the variation in altitude beneath that feature class, however the range in land-uses undertaken by that one farm tenant are likely to be just as great, and therefore even at this small scale %SOC variation can be large. The application of mean values collected from national databases such as the CSS or NSRI will result in even less accurate %SOC estimates as a result of the values being taken from an area of a much greater scale (national level), increasing the likelihood of an even greater range in altitude and other possible controlling factors of %SOC beneath that one land-use or soil group/series etc. This study reveals that for the Wallington Estate in north east England the most accurate estimates of %SOC for particular locations are made when mean values taken from the same particular land-use/soil series/altitude/farm tenancy combination as that of the area in question are applied. Table 2.3 emphasises how the application of these mean values may be responsible for the increase in predictive accuracy due to the much smaller scale of the estate covered by a land-use/soil series/altitude/farm combination compared to the scale of the Estate covered by individual factors such as major soil group.

Although it was earlier suggested that soil series is a better predictor of %SOC than land-use, this was based on single variance analysis and is likely the result of soil series having a smaller variation in altitude and pH than land-use (Table 2.3). When altitude and pH are constant, land-use has been identified as a better predictor of %SOC than soil series, but more importantly farm tenancy is also a better predictor than soil-series. This is confirmed by the greater magnitude of the effect of farm tenancy (35.29%) compared to that of soil series (9.17%) and land-use (19.67%) when all other variables are constant. This research suggests that different farm management practices within a land-use category are causing differences in %SOC, and therefore that land-use stratification into the categories arable, improved temporary pasture, improved permanent pasture, forestry and woodland is not sufficient on which to base SOC baseline estimates.

Table 2.2 shows the variation in %SOC that can be explained using different combinations of the variables discussed here. Comparison of the bottom 4 rows indicates that farm tenancy is an important variable to include and emphasises the suggestion that farm management practices are controlling %SOC. Examination of Table 2.3 also reinforces this suggestion. Although the scale of the land area from which the mean %SOC is calculated has declined when stratification of soil series/land-use/altitude classes is increased into soil series/land-use/altitude/farm tenancy classes, there is no decline in the number of other possible controls on %SOC. It must therefore be either land-management differences between farms, or some other unidentified factor which also varies under different farms that is responsible for the observed variation in %SOC.

Although this research highlights the importance of including farm management practices in any SOC predictions, it has not identified what precise farm management practices are responsible for increasing SOC levels. It has been suggested that fertiliser use can cause a loss of CO_2 to the atmosphere (Zhang and McGrath, 2004), however this is not taken into account when predicting SOC baselines and is an area needing further research. Although many attempts at predicting SOC baselines have stratified the areas into land-use, recognising a difference between improved and unimproved agricultural grassland, this study suggests that this stratification does not go far enough, and that factors such as fertiliser type and application rates as well as grazing intensity and type may be other factors playing a major role (Soussana et al., 2004). Sonneveld et al. (2002) also recognise the need for further research into this area, quote: "Distinguishing between mowing and grazing regimes or specific silage maize cultivation practices might further explain the variability observed." Much recent literature has attempted to establish the role of fertiliser input on SOC stocks (Triberti et al., 2008; Purakayastha et al., 2008), however these factors are rarely considered when establishing SOC baselines. Dedonker et al. (2004) reveal that organic amendments increase soil carbon levels. A lack of disturbance reduces outputs. Previous studies have found animal manure incorporation to increase carbon accumulation, as well as sewage sludge incorporation, straw incorporation and no-till management. These previous findings combined with the results of this study go towards further confirming that SOC is affected by agricultural management (Frazluebbers and Stuedemann, 2008) due to changes in the levels of OM input and soil disturbance. Crop type, crop rotation, tillage type, fertiliser use and organic amendments all influence the amount and distribution of the OM within the soil. Management practices also influence how OM is lost as a result of soil erosion, plant harvest and microbial decomposition. Differences in %SOC between farms located at similar altitudes and on similar soil types at Wallington help to emphasise that land management practices such as these have a large impact on SOC levels. Despite this realisation, farm management is often ignored when predicting SOC levels for un-sampled regions. Franzluebbers et al. (2001, Cited in: Franzluebbers and Studmann, 2008) found greater SOC accumulation in pastures that were grazed by cattle in summer compared to those that were not grazed. In other studies however, no differences have been found between lightly grazed and un-harvested grasslands- but differences have been found between those that are heavily grazed and un-harvested. The results are very mixed but there is clearly a difference resulting from management practices, supporting the results of this research, confirming that SOC baselines and estimations without consideration of these factors will be inaccurate.

The use of secondary data to estimate a SOC bank has many limitations. Although the %SOC values estimated for the estate using NSRI mean values for soil group or soil series can explain approximately the same amount of variation **in** %SOC values as using the mean values collected in the field (16.85% using NSRI major soil group, 16.18% using fieldwork major soil group, 48.35% using NSRI soil series and 48.69% using fieldwork soil series), examination of maps produced by these approaches (Figure 2.3) reveal many spatial differences in the %SOC values across the estate depending on the source of data. This method of estimation relies on soil survey data measured in the 1980s to calculate the soil carbon stock and could be inaccurate due to land-use change and climate change since the period of survey (Gao et al., 2008). The same inaccuracies are therefore likely in any attempt to calculate a region or organisations carbon stock in Britain using the NSRI database, and could be responsible for the range in estimates of SOC stocks and maps produced in this study. The majority of NSRI surveying was undertaken in the 1970s/1980s and climate/land management change could have resulted in a change in soil carbon values for the same soil types under present day conditions (Bellamy et al., 2005; Smith et al., 2007). This could help to explain the differences in %SOC calculated in this field study and those that would be predicted for the Wallington site using NSRI data from earlier decades.

This study shows that predicting a SOC bank based entirely on %SOC values for soil-type is insufficient. This can be expected as SOC is known to vary greatly as a result of land-use, and therefore to predict a region’s carbon stock using just soil-type mean %SOC values is ignoring this major influence on SOC levels. Prediction at a large scale may be accurate in terms of a figure for total C stock, due to the value averaging out over all land-uses; however this method is unlikely to correctly predict the SOC stock values for particular locations. The assumption that agricultural soils, for example, will have the same SOC values as forestry soils if they belong to the same soil series should not be made (Heath et al., 2002). A large amount of the variability in SOM is unexplained by soil classification (Schulp and Veldkamp, 2008) and this research highlights that soil classification can miss the variation within soil classes.

Although it is suggested that it is land-management practices within a land-use that are responsible for the statisitically significant differences in %SOC between farm tenancies located on the same soil type, at the same altitude and under the same land-use class, the possibility that these differences are the result of issues associated with scale must also be considered. In this study all estimates using soil-type as a SOC predictor, whether it be the use of field data %SOC values, CSS %SOC values or NSRI %SOC values is that they are all estimated using the NSRI soil map. Major errors can occur in extrapolating point data if small inclusions of organic soils occur within a mapped soil unit and these are then either not accounted for (if the sample was not taken from the inclusion), therefore the carbon stock is under-predicted, or the carbon stock may be greatly over-predicted if the representative profile for the soil unit was taken from the inclusion, and this value is then applied to the whole soil type. The larger the scale of the soil map, the more errors in carbon inventories (Arnold, 1995), however these limitations are very difficult to overcome as these maps provide the most accurate identification of soil type if extensive sampling is not to be carried out. It is possible therefore that some of the difference that appears to result from farm tenancy could in-fact be the result of inaccurate soil series allocation due to the use of a 1: 50 000 scale soil map.

Other possible explanations for the apparent role of land-management in this study are related to aggregation issues. In this study the low predictive value of using land-use data alone could in part be explained by the subjective nature of classifying particular land-uses. This again however emphasises the role of farm management and stresses the fact that levels of management within a land-use category are an important control of SOC levels. It is very possible that the SOC predictions would be different had a different land-use map been used (Meersmans et al., 2008). The apparent differences between farm tenancies located on the same soil series at the same altitude and under the same land-use could therefore possibly be the result of a particular land-use under one farm tenancy being allocated a different/same land-use to the same/different land-use under a different farm tenancy due to the subjective nature of classification.

Although the application of mean values from local sampling rather than mean values from National databases appears the more appropriate method for SOC baseline estimation, the time and effort involved in such an intense soil sampling campaign must be considered. As the results from this study are presently only valid for the Wallington Estate the mean %SOC values for particular land-use/soil series/altitude locations can not yet be applied to other areas of the country, however ongoing validation studies in these areas will reveal if this can be the case in the future. In order for an organisation such as The NT to estimate their entire SOC stocks it would therefore be most beneficial to use national databases, provided that the soil data is adjusted to take account of land use and altitude using similar correction factors as found in this study. The previous suggestion however, that %SOC values from national databases may now be inaccurate due to the passing of several decades since data collection means that if values from this current study can be found to correctly predict the %SOC in other NT estates then referral to this database should be the method employed in the future. The implication from this study that farm management practices are responsible for differences in %SOC also suggests that in the future the mean %SOC values from national databases could be increased or decreased to take account of practices such as fertiliser application rates and grazing levels, however to date these adjustments cannot be made until the exact effects of land-management on %SOC are clarified.

It must also be realised that this study has only assessed the accuracy of SOC baseline estimates made by aggregating %SOC values from a variety of soil types and land uses from national databases and local soil sampling into different classes to produce mean %SOC values which are then applied to the area of that classification. The study has not assessed the accuracy of SOC baselines produced using process models and geo-statistical methods.

2.5 Conclusion

Calculating a SOC baseline based on major soil group stratification is the least accurate method and is significantly improved by stratification into soil series. Land-use stratification is a less accurate method than soil series; however this can be improved by stratification into soil series/land-use combinations.

Intensive soil sampling at Wallington, north east England has shown that other variables must be included to increase this accuracy further, and that the use of secondary data is insufficient if the most accurate SOC bank estimates are required. The results of this study can be summarised as follows:

- An increase in the ability to explain %SOC variance from 16.85 % to 48.35 % when using soil-series rather than major soil group NSRI %SOC values indicates that if NSRI data is the only data available then this form of stratification should be used.
- Additional information including altitude and soil pH is required to produce more accurate estimates, and these can be improved further still if the areas are also

stratified by farm tenancy. This is shown by an increase in the variation explained by soil series/land-use combination %SOC values of 57.72%, to 59.27% for soil series/land-use/pH/altitude combination %SOC values, to 66.65% for soil series/land-use/pH/altitude/farm tenancy combination %SOC values.

- With all of these variables included in an estimate of SOC levels at Wallington, 33.5% of the variation in SOC still remains unexplained.
- This study suggests that stratification into a greater number of land-use categories is needed in order to take account of different land-use management practices within a land-use category, as well as emphasising the large spatial variability in %SOC.
- The current best estimate of SOC stock on the Wallington estate is 785.24 Kt C. This is in comparison to an estimated above-ground biomass carbon stock of 60.29 Kt C, highlighting the huge importance of preserving and correctly managing SOC.

Chapter 3

Land-management and %SOC

3.1 The impact of farm tenancy on %SOC: an introduction

Statistical analysis of the results from 618 mineral and organo-mineral soil samples taken from the Wallington Estate has identified differences in %SOC between farms even when the %SOC of samples has been adjusted to take account of altitude, soil-series, land-use and soil pH (see Chapter 2). This suggests that different forms of land-management used by different farms could be causing differences in %SOC. In relation to pasture land-management, research in recent literature is generally inconclusive regarding the role of grazing regime, with contrasting results presented in different studies (e.g. Garcia-Oliva et al., 2006; Elmore and Asner, 2006; Reeder and Schuman, 2002; Silver et al., 2010). The same degree of uncertainty is also true regarding the impacts of fertiliser and manure use on %SOC levels under pasture (Hassink, 1994). Research into arable farming methods is more conclusive, and the general consensus is that %SOC in these farming systems can be increased by reduced tillage, and the use of cover crops (Sousanna et al., 2010; Franzluebbers, 2005). Analysis of soil samples collected and described in Chapter 2 revealed that the most significant differences between farm tenancies with all other variables held constant occurred on the land-use classified as rough pasture, although there were also statistically significant differences in %SOC between farms for arable land-use.

The aim of this chapter was to identify reasons why farm, as a factor, was found to be a significant control on %SOC under rough pasture and arable land-use as shown in in Chapter 2. The objective of this research was to investigate land-management techniques undertaken on rough pasture and arable land at Wallington, and to attempt to identify a correlation between any of these land-management practices and %SOC. The initial aim was to identify, for each respective land-use, those farms under which %SOC was unusually high, and those farms under which %SOC was unusually low. It was then hoped that information on land-management techniques and procedures provided by farm tenants would reveal a relationship between land-management and %SOC. This information would be collected using land-management questionnaires, although it was understood that this type of investigation would limit the form of data analysis that could be used, and therefore limit the strength of any conclusions on the role of land-management and %SOC. For this reason it was realised that the results of this chapter would act only as a guide to suggest areas for further research, but it was hoped that hypotheses could be formulated relating to land-management techniques and their impacts on SOC accumulation and loss. A literature search would be undertaken initially in order to establish any apparent known land-management controls on %SOC; to identify uncertainties relating to land-management and %SOC, and to assess gaps in knowledge which need further research and clarification. The results from the literature search would then be used to formulate the questionnaires, and as a comparison against which the results from the survey could be compared.

In addition to assessing the role of pasture land-management techniques on %SOC via land-use questionnaires, a soil sampling campaign would also be undertaken to investigate the impacts of grassland cutting techniques. This area of investigation was considered important for this study as the NT stressed that the effect hay and silage cutting had upon %SOC was of particular concern. The NT believed this aspect of land-management should be investigated, as it is a simple land-use change which they can put into place if it is found to bring carbon benefits. The variability in %SOC beneath improved pasture on the Wallington Estate (see Chapter 2) led the NT to question the impacts of such land-management change.

Formulation of the land-management questionnaires, analysis of these results, and any relationships found between land-management techniques and %SOC will be the focus of this chapter, with the purpose of identifying areas requiring further research.

3.2 Land-management and %SOC: a literature review

3.2.1 Grassland land-management

A literature review into the impact of grazing on SOC in mineral and organo-mineral soils reveals many uncertainties, and a requirement for much further research. In relation to grazing intensity Soussanna et al (2010) imply that heavy grazing is detrimental to SOC, and suggest that stocks will increase following a reduction in its intensity. These results are supported in research by He et al (2009), with observations of lower SOC under areas continuously grazed relative to those where grazing has been excluded. Further evidence that overgrazing causes a loss of SOC to the atmosphere is apparent in the literature (e.g. Elmore and Asner, 2006; Snyman and Du Preez, 2005). Although these studies suggest that grazing should be reduced to enhance SOC, conflicting results emphasise the degree of uncertainty, with much research having found the impacts of grazing intensity, or the presence/absence of grazing, to be insignificant (e.g. Silver et al.,2010; Chan et al., 2010; Patra et al., 2008) In many of these studies (e.g. Hassink, 1994) increased PP and animal excreta were expected to cause increases in SOC, however research revealed that these were insufficient to offset losses from increased microbial activity and respiration- all of which were enhanced by grazing. The insignificant effect of grazing observed in other studies (e.g. Golluscio et al., 2009) was observed despite an increase in bare ground and a reduced litter input to the soil. In contradiction to research calling for a reduction in livestock grazing to enhance SOC, some studies have observed increased soil respiration with reduced grazing intensity, or reduced soil respiration with increased grazing (Cao et al., 2002; Davidson et al., 2000 and Owensby et al., 2006). Although these results do not reveal the net effect of grazing on SOC, initial suggestions are that increased grazing intensity can cause a loss of carbon to the atmosphere. Further results to support the suggestion that increased grazing rates can enhance SOC are provided by Reeder and Schuman (2002), the result, they suggest, of increased annual shoot turnover enhanced by grazing. The impact of grazing on mineral and organo-mineral soils is clearly very uncertain, and much more research is needed before any firm conclusions and land-management change can be made at Wallington. As noted by Maia (2008) the interpretation of grazing intensity is very subjective, and could be in part responsible for the variation in results.

The degree of uncertainty discussed here relating to grazing and grassland SOC is also apparent in research regarding fertiliser application and SOC, with a large number of conflicting studies, and others reporting no change. Some research has hypothesised increased SOC as a consequence of greater grass productivity following fertiliser application, and a subsequent increase in grazing and dung and urine inputs (Schipper et al., 2009; Golluscio et al., 2009). It is clear that fertilisation of grasslands can have a direct effect on grazing patterns and intensity, and so the same uncertainties relating to grazing regimes outlined above must also be considered. Research specific to grassland fertilisation undertaken by Schipper et al (2009) found no significant change in SOC with fertilisation, suggesting that if grazing and animal excreta are increased following fertiliser application then simultaneous increases in SOM mineralisation must also have taken place, as argued earlier in regard to grazing and SOC. Although several studies have assessed the impacts of fertilisation on particular SOC emission or sequestration pathways, there is again an issue relating to the requirement to know the net effect on SOC. Research to support the theory that grassland fertilisation can enhance SOC comes from Amman et al (2007), who found a reduction in SOC mineralisation following nitrogen fertilisation, implying therefore that this land-management practice could increase SOC. In support of this research other authors (Gong et al., 2009; Hyvonen et al., 2008) suggest that PP will increase due to nutrient provision, and in combination with greater litter inputs to soil increased SOC will be observed. Despite the suggestions of increased PP and litter input the uncertainty regarding the net effect on SOC is revealed by Gong et al.'s (2009) hypothesis of stimulated microbial activity and increased SOC decomposition following fertilisation. Many theories such as these refer to counteracting effects on SOC and suggest that the overall impacts could be minimal. In addition to gaseous carbon, losses as DOC need also to be considered, however such research specific to grassland fertilisation on mineral soils is severely lacking. Mctiernan et al (2001) observed increased losses of DOC with increasing nitrogen fertiliser application, but the results from only one study can not be used to confidently state that this will be true of all such soils in differing environments. In relation to the specific impact of fertiliser type and amount on SOC, no research was found in the literature. The impacts of grassland fertiliser application are clearly very uncertain (Hassink, 1994) and can vary considerably between sites (Wang and Fang, 2009). For this reason much further research is needed to find a link, before any land-management changes to increase SOC stocks can be made with confidence at Wallington.

3.2.2 Arable land-management

As indicated in Section 3.1 there is a much stronger consensus concerning the impacts of arable land-management techniques on SOC compared to that of grassland management, with stubble retention and soil tillage often reported to be two of the greatest impactors.

The impact of stubble retention has been well-researched, and with the greatest source of OM supplied to soils being that of crop residues (Havlin et al., 2005) maintaining these residues has been shown to promote SOM and is often encouraged. In contrast, the burning of crop residues in the past has been shown to decrease carbon inputs (Havlin et al., 2005) and has therefore been discouraged. Studies and research in the literature regarding soil tillage is also in good supply. Microbial oxidation of SOM is increased by aeration, and tillage and practices which disturb the soil are known to increase its losses.

Although it is often advised to incorporate crop residues, Havlin et al (2005) suggest that the increased losses due to tillage will outweigh the gains from crop residue, and that carbon gains will therefore be greater if residues are left on the surface. The theory that no-till management will increase SOC is supported by many authors (Franzleubbers, 2005; Sousanna et al., 2010; Ovando and Capparos, 2009, Liu et al., 2009), with reports of increases in SOC of 57 g $C/m^2/yr$ following conversion from plough to no-till management. The reason for increased SOC upon conversion from conventional till to no-till is often attributed to its protection from rapid oxidation and the formation of soil aggregates (Puget and Lal, 2005).

In relation to fertiliser application, an adequate supply is shown to increase the inputs, and retain the cycling of crop residues and plant roots through the soil (Schjonning et al., 2007). Persson and Kirchmann (1994) state that SOC increases following mineral nitrogen fertilisation as a consequence of increased stubble and root formation, and the consequential increase in the material available for humus formation. Gong et al (2009) also found the application of mineral fertiliser to increase SOM, attributing this to greater crop productivity resulting from nutrients added in the fertiliser over-riding any increased decomposition which could occur as a result of fertiliser stimulating microbial activity. A modelling study by Tan et al (2009) showed an increase in SOC with increasing nitrogen fertilisation, the result of reduced SOC emissions and increases in crop productivity. Although the application of fertiliser appears to be beneficial in relation to un-treated land, Roupp (2001) found that the amount of fertiliser applied had no effect on SOM, nor did the impact of fertiliser on SOM comply with the impacts on crop yield. The application of mineral fertiliser resulted in a yield of Winter Rye 33% higher than that when no fertiliser had been applied, and for potatoes a yield 10% higher. This emphasises that the impacts on all aspects of the carbon balance must be assessed to avoid making false assumptions regarding fertilisation; an increase in crop productivity does not necessarily correspond with an increase in SOC.

Manure incorporation into arable land is another well researched area of land-management effects on SOC; however the results are still uncertain and require further work. Jones et al (2005) observed much greater soil respiration from plots receiving poultry manure than those left un-treated, suggesting potential declines in %SOC. In contrast, other research reports a potential for carbon sequestration following manure incorporation (Ovando and Caparros, 2009; Gong et al., 2009; Schjonning et al., 2007), with findings of rapid increases in SOM and SOC in time periods as short as 5-6 years. These results are not however supported by Cuvardic et al (2004), who found no significant change in SOC following treatment. The impacts of manure application are clearly uncertain, with the uncertainty relating not only to manure application versus no manure application, but also to the type of manure applied.

Literature searches also suggest that SOC under arable land-use could be influenced by the type of crop grown and the crop rotation system, with reports of higher respiration under row crops than cereals (Rees et al., 2005; Kasimir- Klemedtsson et al., 1997) A diverse crop rotation system is reported by Schjonning et al (2007) to increase SOC, with other reports of increases under fields subjected to arable/grass ley rotations compared to those growing continuous row or cereal crops (Christensen, 1998: Curvardic et al., 2006). Investigation by Franzluebbers (2005) showed significantly higher SOC under fields with cover crops in their rotation cycle, and land left fallow is often shown to decrease in SOM, as is land under low crop productivity (Havlin et al., 2005).

The relative importance of different management impacts on arable SOC is also uncertain, and requires further research, with Christensen (1998) finding greater increases in SOC from manure application than those from converting to a arable/grass ley rotation system. Several studies have found greater SOM in topsoil under fields applied with manure compared to those treated with mineral fertiliser, thought to be the result of reduced carbon losses (Roupp, 2001) or humus accumulation (Drinkwater et al., 1998). In relation to the relative impacts of manure and straw residue incorporation manure is reported to be less easily decomposable, having already degraded during storage, a possible reason for the higher SOC observed under land treated this way by Persson and Kirchmann (1994). Conversion to complex cropping systems is reported by Jarecki et al (2005) to sequester less SOC than a conversion in tillage techniques, and a further uncertainty relates to the impacts on SOC when the land-management techniques discussed are used in combination. The greatest increases in SOM have been found to occur when a combination of mineral fertiliser and manure are applied (Rudrappa et al, 2006 Cited in: Gong et al., 2009).

3.2.3 Grassland cut for hay or silage

A review of literature has found no information relating to the impacts of hay and silage cutting on %SOC.

3.3 Materials and methods

3.3.1 Study site

This chapter analyses the results of soil samples taken from all the farm tenancies on the NT Wallington Estate (displayed in Figure 1.3).

3.3.2 Statistical analysis

The database of 618 mineral and organo-mineral soil samples collected from farms on the Wallington Estate was split into individual land-use classes. ANCOVA was undertaken on each respective land-use dataset using the same statistical methods as in Section 2.2.4. The name of the farm from which each soil sample was collected was recorded along with its soil series, altitude and soil pH. The categorical variables 'farm tenancy' and 'soil series' were entered into the GLM as factors, and 'soil series', 'altitude' and 'soil pH' as covariates., ANCOVA was then undertaken to identify which farms had significantly different ($p<0.05$) %SOCs under each respective land-use, when controlled for the other factors and covariates. A main effects plot was then generated to reveal the mean %SOC under each farm, when controlled for the other factors and covariates. The mean %SOC values from each farm were then compared, allowing identification of the farms under which %SOC was unusually high/low for this land-use. These mean values are calculated from the following number of samples taken from rough pasture: Catcherside: 23; Chesters: 10; Donkin Rigg: 26; Dyke Head: 5; Elf Hills: 6; Fairnley: 7; Fallowlees: 6; Gallows Hill: 27; Greenleighton: 19; Harwood Head: 19; Newbiggen: 7; Ralph Shield: 20; Rothley West Shield: 25; Rugley Walls: 4; Tuthill: 37 (where sample numbers are relative to the area of rough pasture on each farm). The raw data relating to %SOC and farm location is provided in Appendix 2 under the heading 'Wallington soil samples'.

3.3.3 Land-management questionnaires

The clear uncertainty in the literature, and a lack of any conclusive evidence from Chapter 2 concerning land-management techniques and variation in %SOC meant that no immediate land-management change suggestions could be made. It was also considered unbeneficial to undertake any land-management intervention trials at this stage of the research, as this was thought unlikely to produce any clear conclusions due to the short period of time available over which to assess any changes, and the shortage of current knowledge on what exactly could result in beneficial carbon gains. Instead it was thought more beneficial to extract this information by looking in detail at the variation in land-management practices between farm tenancies on the Estate, and to see if a relationship could be found between land-management and SOC. If any potential land-management impacts are apparent following this study a land-management study could then be undertaken, to verify or falsify any formulated hypotheses. In an attempt to identify what could be causing the differences in %SOC between farm tenancies at Wallington, land-management questionnaires were developed for those land-uses under which %SOC differed significantly by farm: rough pasture and arable.

The questionnaires were constructed to try to identify general land-management practices within a farm tenancy as a whole, rather than looking for field by field land-management practices. The questions for each farm tenancy were however specific to general land-management of fields from which the soil samples were taken. The purpose of the questionnaires was to try to identify any similarities and consistencies in land-management practices between the farms ranked together at either end of the %SOC scale for that land-use.

Within the land-use 'rough pasture' the aim was to identify any consistencies in grazing regime, fertiliser use, and manure application between tenancies with similar SOC levels, and any major differences in these land-management practices under farms that had very different %SOCs. Attempts were made within this land-use to resolve the following questions:

- Does the % of fields grazed by sheep and/or cattle vary with farm, and is there a difference in %SOC caused by this variation in sheep and cattle grazing?
- Does the average stocking rate and weight/breed of livestock vary with farm, and does this have an impact on %SOC?
- Does the length of livestock grazing and the time of year of livestock grazing vary with farm, and are there apparent differences in %SOC between farms which graze their stock for longer periods, or at different times of year?
- Does the amount and type of manure application vary with farm, and is there a difference between farms which apply manure and those which don't, or between those which apply different types of manure, at different frequencies, or at different times of year?

- Do all farms apply fertiliser (mineral or organic), and is there a difference in %SOC between farms which apply fertiliser and those which don't?
- Is there a difference in the amount and type of fertiliser applied by farms, and does this correlate with a difference in %SOC?
- Do farms differ in their method, frequency and timing of fertiliser application, and does this have an impact on %SOC?

An example questionnaire constructed to help answer the questions relating to rough pasture land-management is provided in Appendix 3.1.

Within the arable land-use category the intention was to identify any consistencies in tillage methods, crop-types and manure and fertiliser application between farms with similar %SOC levels, or major differences in these land-management techniques between farms with very different %SOCs. For arable land-use it was hoped to answer the following questions:

- Does the number of fields continuously cropped, and the number in rotation with pasture, cover crops or fallow periods vary with farm, and do these variations correlate with a variation in %SOC?
- Does the % of fields planted with row crops, cereal crops and legumes vary with farm, and does this translate to a variation in %SOC?
- Are there any significant differences in the length of the crop cycle and the timing of planting and harvest between farms, and is this responsible for any observed variations in %SOC?
- Are there differences in tillage regime and the treatment of crop residue between farms, and does this correlate with %SOC variation?
- Are there variations in the type, amount and frequency of application of manures and mineral fertilisers, and do these help explain the variation in %SOC with farm?

An example questionnaire constructed to help answer the unknowns related to arable land-use is provided in Appendix 3.2.

3.3.3.1 Compiling and interpreting questionnaire results

The results provided by the farmers relating to fertiliser application rates were given in a variety of formats including hundredweight/acre, kg/hectare and kg/acre. In order to make comparisons, all results were converted to kg/m^2. In relation to the amount of fertiliser applied, the results from the farmers were provided in respect to the total weight of fertiliser rather than the weight of each nutrient. Typical nutrient compositions of particular fertilisers had consequently to be estimated based on a review of the literature and reference to fertiliser information, or by information provided directly by the farmers. The total weight of each nutrient applied by each farmer was calculated based on the % of each nutrient within the fertiliser, and the weight of total fertiliser applied. Some fertilisers were applied annually and others on a 4 year rotation. It was thus considered best to calculate the total fertiliser application over a 4 year period and then convert this to an annual value. In regard to rough pasture application a variety of fertilisers were reported to be in use, ranging from nitrogen based, to compound, to phosphate-based fertilisers. In this study the following fertiliser compositions were assumed:

- Basic slag (a phosphate based fertiliser): phosphate 15% (Benton-Jones, 2003)
- Gafsa (a rock phosphate): 12.5 - 40% (Ankomah et al., 1995; Scholefield et al., 1997; Collings, 1955; Havlin et al., 2005)
- Monoammonium phosphate: 48 – 62% phosphate, 11 – 13% nitrogen (Havlin et al., 2005)
- Ammonium nitrate: 34% nitrogen (information provided by farmer)
- NPK compound fertiliser: 20% nitrogen, 10% phosphate, 10% potassium (Information provided by farmer)

Using this information calculations were made to establish the weight of each nutrient applied to each farm's rough pasture. The calculations can be found in Appendix 4.1. Greenleighton, Dyke Head, Fairnley, Fallowlees and Harwood Head farms applied no fertiliser to any of their rough pasture fields, and although Catcherside farm applied phosphate fertiliser at a very low rate more than six years ago, it has not applied any in the last six years therefore its application rate was considered as zero in this study. No information on fertiliser application was provided by Ralph Shield and Elf Hills farms, therefore neither farm could be included in the analysis. In regard to fertiliser application to arable land Newbiggen farm applied ammonium nitrate fertiliser and this was assumed to have the same composition as that applied to rough pasture at Gallows Hill. A compound fertiliser with the composition 0:20:30 was also applied at Newbiggen (information provided by the farmer). Donkin Rigg farm applied nitrogen fertiliser, and given the lack of detail provided relating to its composition it was assumed to be ammonium nitrate of the same nitrogen content as that applied at Newbiggen. As with rough pasture this information was used to calculate the weight of each nutrient typically applied to arable fields on each farm. The calculations can be found in Appendix 4.2.

The results provided by farmers relating to sheep and cattle stocking rates were generally given as number of livestock/acre. As it was considered a possibility that livestock weight and the length of the grazing season could also be responsible for variations in %SOC it was decided best to multiply the stocking rate by the weight of the livestock and the number of days of livestock grazing. This resulted in stocking rates represented as $Kg/m^2/yr$.

3.3.3.2 Analysis of questionnaire results

Once all questionnaire results had been compiled and converted into comparable formats, as in Section 3.3.3.1, the data was entered into Microsoft Excel and Minitab statistical software. Statistical analysis by way of regression analysis was undertaken on the 'rough pasture' %SOC and land-use data to quantify how much variation in %SOC could be explained by the land-management practices undertaken by each farm. Although the mean values are calculated from many soil samples, the nature of analysis (with adjustment for uncontrolled factors and covariates) means that uncertainty ranges and error bars cannot be displayed in the following regression analysis. Regression analysis was undertaken by plotting the results from the land-management questionnaires against the mean farm %SOC values i.e. when investigating the impact of fertiliser application on %SOC the weight of fertiliser was regressed against the mean %SOC of each farm. Data from the arable land-management questionnaires could not be assessed statistically as only two farms provided the necessary information.

3.3.4 Hay versus silage trial

In an attempt to establish whether hay or silage cutting has an influence on %SOC comparisons of the %SOC of soil samples from both field types was undertaken. Due to the potential influence of land-use history on %SOC (see Chapter 2) it was necessary to compare fields which had been in their respective land-uses for several years. Consultation with the NT Estate warden and the Wallington Estate biological survey revealed only one such field for each land-use on the Estate: a hay field located at Rothley West Shield and a silage field located at Dyke Head. Both fields were positioned at similar altitudes- therefore reducing the number of variables in the experiment. The soil series present in both fields were however different, with the field at Dyke Head overlying three individual soil series: Brickfield, Dunwell and Waltham, compared to only Brickfield at Rothley West Shield. 30 soil samples from each field were taken from a depth of 20cm (approximate plough depth) in February 2010. Variations in topography within both fields meant that a stratified random sampling technique was adopted for soil sample collection from each field. Soil samples were collected and analysed for %SOC following the methodology described in Section 2.2.2.1.

3.3.4.1 Analysis of hay and silage results

The %SOC values of samples collected from the hay and silage fields were entered into Minitab along with information relating to the field type from which they were taken. As soil series and altitude were not recorded upon soil sample collection these covariates could not be included in any analysis. A one-way ANOVA was undertaken to reveal any significant differences in %SOC with field type.

3.4 Results

3.4.1 Rough pasture land-management

Raw data relating to the questionnaire results and farms %SOC is included in Appendix 5 under the heading ‘Rough pasture Q results’.

3.4.1.1 %SOC under individual farms

Statistical analysis of all soil samples taken from rough pasture when controlled for soil series, altitude and soil pH revealed mean %SOCs under farms in the order of (from high to low): 1.Donkin Rigg: 7.59 %, 2.Rothley West Shield: 7.36 %, 3.Greenleighton: 5.51 %, 4.Newbiggen: 5.38 % , 5.Dyke Head: 5.25 %, 6.Tuthill: 5.02 %, 7.Rugley Walls: 4.81 %, 8.Fairnley: 4.39 %, 9.Chesters: 4.37 %, 10.Gallows Hill: 4.29 %, 11.Catcherside: 4.07 %, 12.Harwood Head: 3.55 %, 13.Fallowlees: 2.28 %. (As Elf Hills and Ralph Shield farms did not provide land-management information they could not be included in the analysis and thus were not ranked).

3.4.1.2 Questionnaire results: Fertiliser application

The results of the questionnaires revealed that the use of fertiliser on rough pasture was not consistent across farms on the Wallington Estate, with variation in both the amount and type of fertiliser applied. There were however consistencies in the methods of fertiliser application, with all farms using a broadcaster or similar form of fertiliser spreader. There were also consistencies in the timing of fertilisation, with all farms spreading their fertiliser in the spring months of March, April or May.

In relation to the total amount of nitrogen, phosphate and potassium applied to rough pasture, farms applied fertiliser in the following order: Rugley Walls: 9.7 g/m^2/yr, Donkin Rigg and Rothley West Shield: 9.3 g/m^2/yr, Tuthill: 8.3 g/m^2/yr, Chesters: 3.7 g/m^2/yr and Gallows Hill: 2.1 g/m^2/yr. All other farms applied no fertiliser to their rough pasture. Regression analysis of the amount of total fertiliser applied versus the mean %SOC from each farm revealed that 37 % of the variation in %SOC could be explained by total fertiliser application rate, with %SOC increasing with increasing fertiliser application.

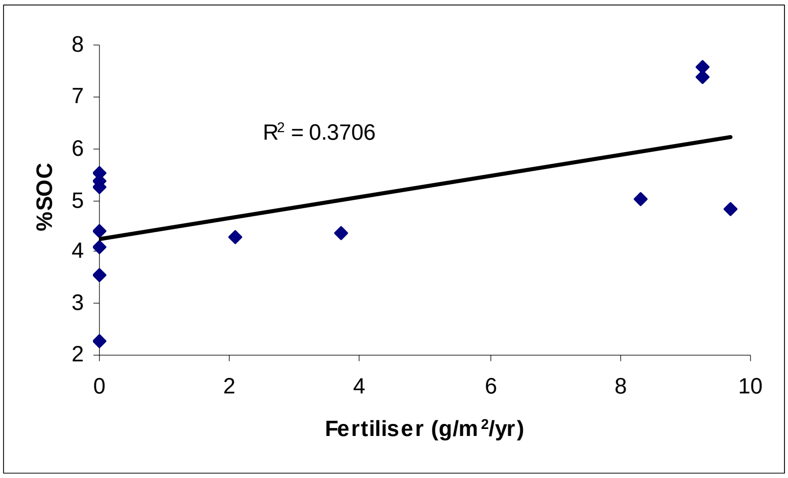

Figure 3.1 The relationship between mean farm %SOC and total fertiliser application

Results from the questionnaires revealed that the farms with the highest and second highest mean %SOC applied the same phosphate based fertiliser (basic slag) and amounts (9.25g phosphate/m^2/yr) to all of their rough pasture fields. This fertiliser contained no nitrogen or potassium and the results led to an investigation into the impacts of phosphate fertiliser on %SOC, displayed in Figure 3.2.

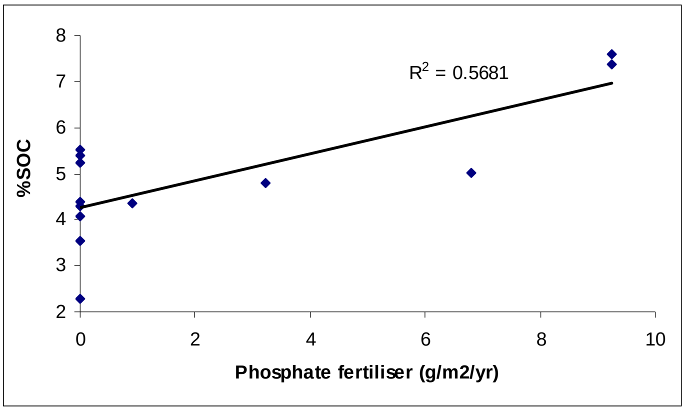

Figure 3.2 The relationship between farm's mean %SOC and the total amount of phosphate applied in fertiliser.

Although the farms with the 3rd, 4th, 5th, 8th, 10th 11th, 12th and 13th highest mean %SOC did not apply any phosphate to their rough pasture, these results do indicate a possible role of phosphate fertiliser in enhancing %SOC. This is confirmed by regression analysis, which indicates that 56.8% of the variation in %SOC could be explained by the variation in the amount of phosphate fertiliser, with %SOC increasing with increasing amounts of phosphate.

In relation to total nitrogen fertiliser the results from the questionnaires are presented in Figure 3.3, and indicate no apparent relationship between nitrogen fertiliser and %SOC. This is confirmed by regression analysis, indicating that less than 1% of the variation in %SOC can be explained by variation in the amount of nitrogen fertiliser. .

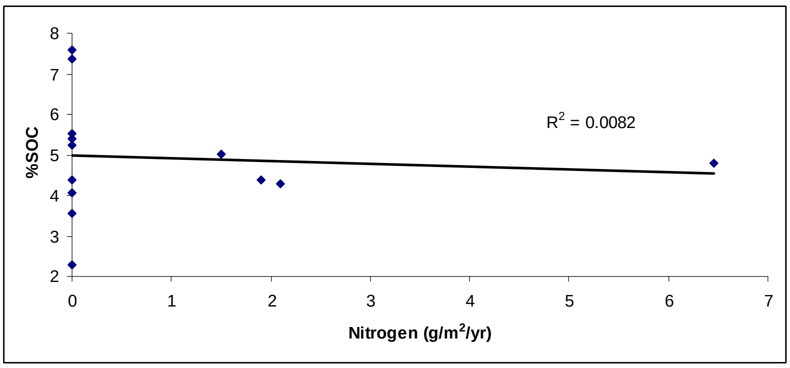

Figure 3.3 The relationship between farm's mean %SOC and the total amount of nitrogen applied in fertiliser.

3.4.1.3 Questionnaire results: Grazing regimes

Analysis of grazing regimes revealed no apparent association between the breed of cattle and the mean %SOC found on farms. 7 of the 13 farms had Aberdeen Angus cattle: those being the ones with the 1st, 2nd, 3rd, 7th, 8th, 10th and 12th highest mean %SOCs. Other breeds of cattle consisted of Continental Cross, Limousin Cross and Beef Shorthorns. The farm with the lowest %SOC was the only farm to have Galloway cattle; however this can not be used to explain the low %SOC as much more evidence would be required. When the impact of sheep breed on %SOC was investigated, as observed with cattle breed, there appeared to be no significant relationship. 75% of their sheep population consisted of North of England Mules, and 25% of Swaledales on the farms with the 1st and 2nd highest mean %SOC. The sheep population of the farm with the 3rd highest mean %SOC consisted completely of the Blackfaced breed, as was the case on the farms with the 8th, 11th and 12th highest mean %SOC. The farms with the 4th and 5th highest mean %SOC consisted completely of North of England Mules, and 50% of the sheep population on the farms with the 6th and 7th highest mean %SOC consisted of North of England Mules and 50% of Suffolk Cross. The farm with the 9th highest mean %SOC had a combination of North of England Mules and Texel cross, and the farm with the 10th highest mean %SOC, of Blackfaced and North of England Mules. The sheep population of the farm with the 13th highest mean %SOC was made up entirely of Cheviot sheep (Figure 3.4).

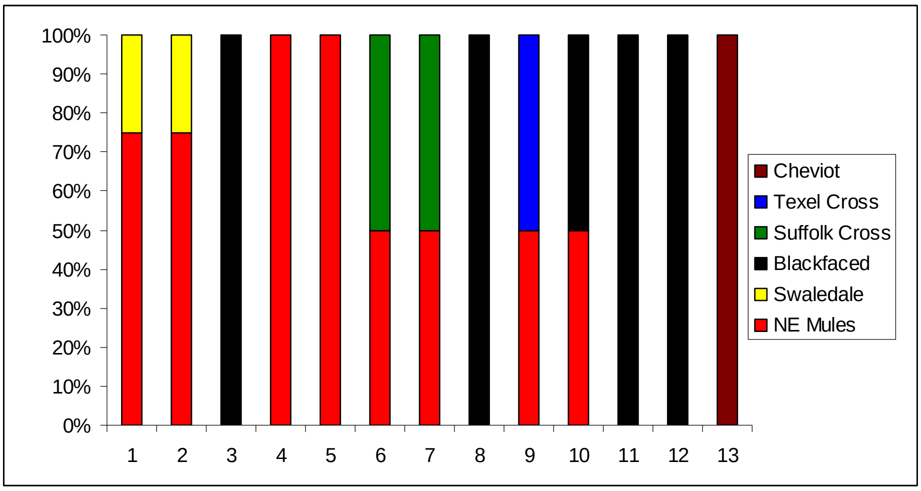

Figure 3.4 Variation in sheep breed and farms ranked by mean %SOC

Of the 13 farms there were only 2 that had fields grazed purely by sheep. Greenleighton (3rd highest %SOC) grazed its fields for 10 months of the year and Newbiggen

(5th highest %SOC) for 6 months. The remainder of the farms grazed all of their fields with a mix of sheep and cattle. The farms with the 1st, 2nd, 6th, 8th, 9th, 10th, and 12th highest %SOC grazed these fields for 12 months of the year with sheep. This is in comparison to the farm with the 3rd highest %SOC, where fields were grazed from January to October, 4th from August to November, 7th where sheep did not graze in April, and 12th where sheep were grazed from only May to July. The farm with the 13th highest %SOC grazed its sheep for only 2 months of the year, however information regarding which months was not provided. The farms with the 1st, 2nd, 3rd and 8th highest %SOC grazed their fields with cattle from May to November, 4th and 12th from July to September, 6th, 7th and 11th from May to December, and 9th and 10th from May to October. The farm with the 13th highest %SOC grazed its fields with cattle for 2 months of the year, but again information was not provided as to which 2 months.

In relation to sheep stocking rate there appeared to be no correlation between total weight of sheep (Kg/m^2/yr) and %SOC (Figure 3.5). This is confirmed by regression analysis, indicating that only 1.3 % of the variation in %SOC can be explained by the variation in sheep stocking rate. In relation to cattle stocking rate there again appeared to be no relationship between the total weight of cattle grazing and %SOC (Figure 3.6). Statistical analysis again confirms this, with an r^2 of 0.023 indicating that only 2.3 % of the variation in %SOC can be explained by the variation in cattle stocking rate.

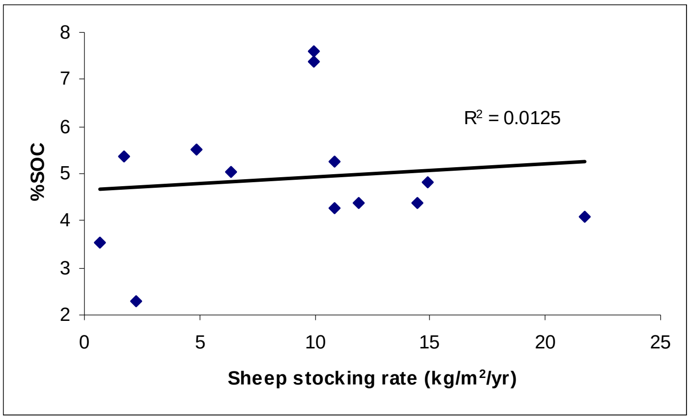

Figure 3.5 Relationship between farm's mean %SOC and total annual sheep stocking rate.

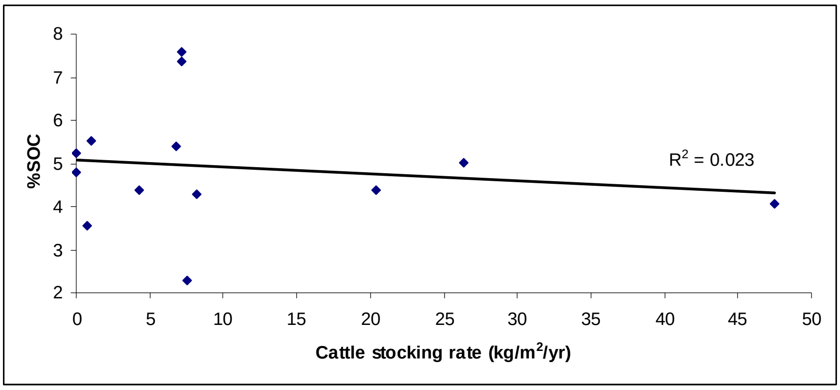

Figure 3.6 Relationship between farms mean %SOC and total annual cattle stocking rate.

In situations where fields were grazed by a combination of both sheep and cattle no correlation between the total weight of livestock and %SOC was again observed (Figure 3.7). Regression analysis of the data revealed that less than 1% of the variation in %SOC can be explained by the variation in sheep and cattle stocking rate.

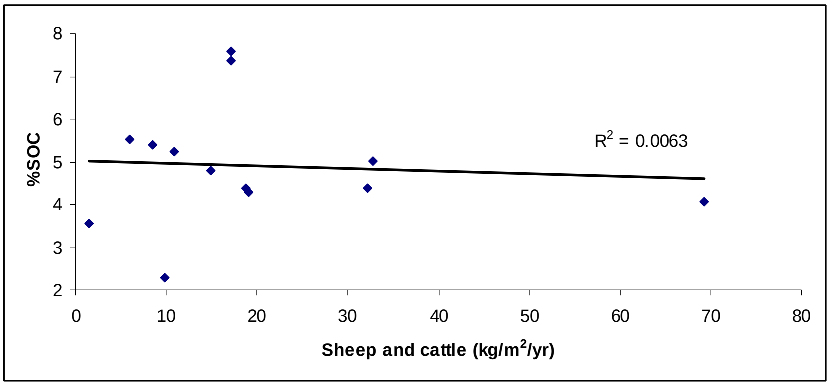

Figure 3.7 Relationship between farms mean %SOC and total annual livestock stocking rate.

3.4.1.4 Questionnaire results: Manure application

The impacts of manure application on rough pasture could not be assessed, as no farm on the Wallington Estate applied manure to grassland in this land-use. Manure application could therefore be ruled out as a factor responsible for causing any of the variation in %SOC on rough pasture on the Wallington Estate; however its true impact on %SOC remains unknown.

3.4.2 Arable land-management

Raw data relating to the questionnaire results and farms %SOC is included in Appendix 5 under the heading 'Arable questionnaire results'.

3.4.2.1 Farms and %SOC

Statistical analysis of all soil samples under arable land-use when controlled for soil series, altitude and soil pH revealed that the soils with the highest %SOCs were found on Newbiggen Farm, followed by Broomhouse and Prior Hall, with the lowest %SOCs under Donkin Rigg. The only statistically significant difference, however, was that between Newbiggen and Donkin Rigg. With neither Broomhouse nor Prior Hall farm participating in the questionnaire, analysis could only be undertaken on the results from Newbiggen and Donkin Rigg farms.

3.4.2.2 Questionnaire results: Cropping systems

In relation to the % of arable fields under a continuous cropping regime the farm with the highest %SOC (Newbiggen) had 50% compared to the farm with the lowest %SOC (Donkin Rigg), with 0%. The remainder of the fields on each farm were in rotation with cover crops. Neither farm had any fields from which samples were taken that were in rotation with grassland or fallow. Newbiggen farm had 50% of its fields in cereal crops (Winter Oats) and 50% in legumes (Red Clover), compared to Donkin Rigg which had 100% in cereal crops at the time of sampling (Spring Barley, Winter Wheat and Winter Barley). Although Donkin Rigg currently has no fields planted with legumes these are part of the rotation system, and when planted are Fodder Rape, Kale or Turnip. The length of the cereal crop cycle at Newbiggen is 2 years followed by 2 years of legumes, and at Donkin Rigg is 4 years followed by 1 year of legumes. The cereal crops at Newbiggen are planted in the autumn, and at Donkin Rigg 4 varieties are planted in winter and 1 in spring. Legumes are planted in the spring at Newbiggen and in the summer at Donkin Rigg. Cereal crops at both farms are harvested in the autumn; however the legumes at Newbiggen are cropped and mulched 3 times in the summer compared to Donkin Rigg where legumes are harvested in the winter. Conservation tillage is undertaken on 0% of the fields at Newbiggen compared to Donkin Rigg where it is undertaken on fields once every 5 years.

Crop residue is removed from 0% of the fields at Newbiggen and from 100% of the fields at Donkin Rigg.

3.4.2.3 Questionnaire results: Manure and fertiliser application

The farm with the highest %SOC (Newbiggen) is reported to apply manure to none of its fields compared to the farm with the lowest %SOC (Donkin Rigg), where 25 t/ha of manure (farmyard muck) is applied annually in September. 17% of fields at Newbiggen are treated with fertiliser compared to 100% at Donkin Rigg, both of which fertilise on a twice yearly basis. The fields to which fertiliser are applied at Newbiggen are treated with 14g nitrogen/m^2/yr, 5g phosphate/m^2/yr and 7.5g potassium/m^2/yr. At Donkin Rigg the fields are fertilised with 2.7g nitrogen/m^2/yr.

3.4.3 Hay versus silage trial

Analysis of all 60 soil samples has revealed a significantly higher SOM and %SOC for the silage field compared to the hay field. These results however are not conclusive as the difference in soil series with field could be playing a significant role in %SOC. Much further research is needed before any conclusions can be made.

3.5 Discussion

Of all the results relating to rough pasture land-management impacts on %SOC the strongest relationship found between land-management and %SOC was that of phosphate fertilisation. All other relationships between variables and farm's mean %SOC were very weak. These results correspond with those in the literature, indicating a large amount of uncertainty and variation regarding grassland land-management and %SOC.

In relation to the impact of stocking rate and general grazing regime on %SOC there was no relationship. The use of quantitative measures for grazing intensity (g/m^2/yr) means that the concerns of Maia (2008) relating to the subjective nature of ascribing grazing intensity can not be responsible for the lack of an apparent grazing impact in this study. The results of this study support those of Silver et al (2010), Chan et al (2002), Patra et al (2008), Hassink (1994) and Wang and Ripley (1997), all of whom found no significant impacts on %SOC as a result of grazing intensity. The results of this study must however be taken with caution as other variables which could have an impact on grassland %SOC such as fertiliser application were not consistent between sites (Section 3.4.1.2). It is very important therefore to recognise interacting factors when interpreting these results, and to recognise that grazing trials on land where all external variables are constant must be undertaken before the conclusion that grazing intensity has no impact on grassland %SOC can be made. Any conclusions regarding the insignificant impact of sheep and cattle breed on %SOC are difficult to establish as farms were found to vary not only in the breed of livestock, but also in the length of grazing season and stocking rate. It may therefore be that the grazing habits of particular breeds of livestock vary with other breeds, and that these grazing habits could cause variation in %SOC. The same is true of breed of sheep and soil compaction, which may result from variations in livestock weight. Soil compaction may also be affected by the timing of livestock grazing; however the time of year of grazing was shown to have no significant impact on %SOC in this study. Although each of these are factors with the potential to cause variation in %SOC they can not be clarified or falsified in a study such as this, due to the uncontrolled external variables and multiple interacting factors.

The results of this study suggest that the application of phosphate fertiliser may be responsible for some of the greater %SOCs found under some farms relative to others. The type of phosphate fertiliser applied by the farms with the 1st and 2nd highest mean %SOC in this study was basic slag, a fertiliser reported in the literature to have a positive response on crop yields. Although the findings of this study are based on the results of a farm questionnaire, they are supported by the findings of Jackman (1964) and William and Hayes (1990), both of whom observed accumulation of SOC under pastures fertilised with phosphorus. These results, however, are not supported by Schipper et al (2009), who found that although the dry matter inputs increased with phosphorus application this did not translate into increased SOC. Trials looking at the impacts of basic slag on crop yields have found increases in forest yields (Piret, 1991), and pasture yields in Northern Spain were increased by up to 50% following applications of 5000 Kg/ha of Basic Slag. A positive response by crops to Basic Slag application is attributed by Collings (1955) to the presence within basic slag of rare essential elements. Although this is a positive response in terms of crop growth and not soil carbon, it does suggest that soil carbon could increase as a result of increased alive and dead plant matter, and suggests that the findings of this study deserve more research.

In relation to nitrogen application the results of this study imply that nitrogen fertilisation has no impact on %SOC. These findings do not agree with the arguments of

Hyvonen et al (2008), who suggest that the result of a change in the C/N ratio resulting from nitrogen fertiliser application will lead to a decrease in heterotrophic respiration and a simultaneous increase in PP. The results of this study are however supported by a large majority of the literature which refers to the counteracting effects of nitrogen fertilisation. Several researchers have found that although PP will increase with fertiliser application, a simultaneous increase in SOC mineralisation will result in no overall net effect (see Section 3.2.1). The results of this study relating to both phosphate and nitrogen fertiliser application must be taken with caution, due to the same issues regarding external variables and interacting factors discussed in relation to grazing regimes.

With only 2 of the farms on the Estate undertaking arable farming it is not possible to draw any firm conclusions regarding their land-use techniques and %SOC. It is possible however to compare the results from this study with those from the literature presented in Section 3.2.2. In relation to stubble retention the results from the literature suggest that SOC increases with stubble retention, and the results of this study support this suggestion, with the farm with the highest mean %SOC (Newbiggen) retaining 100% of its crop residues, compared to Donkin Rigg where 100% of crop residues are removed. As indicated in Section 3.2.2 there is a strong consensus that conservation tillage methods result in an increase in SOC, however it is difficult to establish the impacts of such management on the Wallington Estate, as although there is a difference in tillage methods between the two farms, with conservation tillage being undertaken at Donkin Rigg, this method is only applied 1 in every 5 years, and any SOC gains which may occur in this 1 year are likely to be counteracted in the following 4 years.

The lack of fields on the Wallington Estate which have been in continuous hay and silage land-use means that the impact of each respective land-use on %SOC is not able to be determined. For such a land-use impact to be revealed a much greater number of fields in each land-use would need to be sampled and the soil series and altitude beneath each land-use would need to be consistent if any firm conclusions were to be reached.

3.6 Conclusions

The results of this study suggest a possible link between rough pasture fertilisation and %SOC, in particular that %SOC increases with phosphate fertiliser application. The phosphate fertiliser used on farms with the highest %SOC under rough pasture land-use was basic slag, suggesting a possible link between its application and SOC accumulation. No correlations were found between nitrogen fertiliser application and %SOC; between grazing intensity and %SOC; or grazing regimes and %SOC. The nature of the study and the number of possible interacting factors means that much more research is needed under controlled conditions before any firm conclusions regarding rough pasture land-management and SOC accumulation can be made.

In regard to arable land-management, the results of this chapter suggest that land-management techniques could be responsible for variations in %SOC observed on the Estate, however the comparison of only 2 farms and their land-management techniques means that the results could not be analysed statistically, and that much more research is needed into the impacts of arable land-management on %SOC.

Although the impacts on %SOC of grassland cut for hay and grassland cut for silage was also investigated the results of this study are severely limited by the lack of fields in each respective land-use, meaning that no firm conclusions can be drawn.

Chapter 4

Fertiliser Application to Pasture: Rothley West Shield Field Trial

4.1 Grassland fertiliser application and SOC: an introduction

Investigation into the controls on and distribution of SOC on the NT Wallington Estate in Chapter 2 revealed a large degree of variation in %SOC between land on different farms undertaking the same land-use. This suggested, as explained in Section 2.4, that individual farm land-management practices such as fertiliser application and grazing regimes could be responsible for some of the variation in %SOC beneath grazed pastures. Further investigation via a land-management questionnaire (described in Chapter 3) indicated that higher %SOC under some areas of pasture could possibly be attributed to the application of phosphate based fertilisers. This hypothesis was however based on results indicating a correlation between high %SOC and phosphate application according to the farm land-management questionnaires (Section 3.4.1.2). The subjective and sometimes qualitative nature of the questionnaire results, and the manner in which information was provided, means that much further quantitative research needs to be undertaken before any firm conclusions can be reached. In contrast to phosphate fertiliser the results of Chapter 3 suggest that nitrogen fertilisation of grasslands has no impact on %SOC.

The results from the questionnaire on rough pasture land-management techniques in Chapter 3 led to formulation of the hypothesis that a particular type of phosphate fertiliser (basic slag) may be responsible for increasing %SOC, with basic slag having been applied to the grasslands of those farms with the greatest %SOC's. Basic Slag is a fertiliser produced as a by-product of the steel industry, with a chemical formula of $[(CaO)_5 \cdot P_2O_5 \cdot SiO_2]$ (Collings, 1955) and a phosphoric acid content of at least 8-10%. The findings of Chapter 3 do not however show conclusively that basic slag promotes SOC accumulation, or that its application is responsible for higher %SOC than would be present if land was to be fertilised with NPK or nitrogen fertiliser, or not fertilised at all. In addition to the subjective and qualitative nature of the questionnaires described above, there were also other un-controlled factors, with a variation in grazing regime and soil type between farms possibly contributing to the observed variance in %SOC. Although ANOVA undertaken in Chapter 2 was able to control for some of this variation in soil type, the variations in grazing regime and other possible unidentified controls on %SOC were not taken into account. For the hypothesis of SOC accumulation under fields fertilised with basic slag to be verified, a controlled experiment was therefore considered necessary, and it is that controlled experiment that is the focus of this chapter.

4.1.1 Chapter aims

The aim of this chapter was to establish the impacts of a variety of fertilisers on %SOC, and to provide quantitative and statistical evidence for these impacts. It was hoped to determine if the findings of Chapter 3 could be made more conclusive, and to verify whether SOC accumulates to a greater extent under grasslands fertilised with basic slag than those fertilised with nitrogen, compound fertilisers, or those left unfertilised.

In line with the aims set out in Section 1.3 the impacts on above-ground carbon stocks in vegetation biomass would also be assessed. Any variations in biomass carbon stocks with fertiliser treatment would allow the impact of fertilisers on the combined above and below-ground carbon stocks to be established, and therefore to guide the NT on the most beneficial fertiliser to apply to grasslands for maximum carbon sequestration benefits.

As the major aim of this thesis was to increase %SOCs, and SOC stocks, this chapter set out to assess whether any differences in %SOC could be identified following one year of variation in fertiliser application. Large spatial variability, time constraints, and difficulties in making soil bulk density measurements (Section 2.2.6) meant that all comparisons of SOC made in this chapter are in relation to %SOC and not SOC stock. It is assumed that SOC stock will respond similarly to %SOC but this assumption must be taken with a degree of caution, as a change in soil quality which may result if fertiliser has an impact on %SOC could itself cause a change in soil bulk density (Agbede, 2010).

Although the main aim of this chapter was to detect any changes in %SOC, the short time period of this trial meant that a different approach to the detection of SOC change was also required. Measurement of the fluxes of carbon from and to the atmosphere, and the balance between these fluxes would therefore be undertaken, to help detect any such changes which could then go on to affect SOC accumulation.

In addition to attempting to change land-management for maximum carbon sequestration, it is also recognised that any land-management change must meet the criteria set out by the NT relating to water quality, biodiversity and fossil fuel use, and their aim to reduce the use of artificial fertilisers that rely on fossil fuel inputs (see Section 1.1.2). A further aim of this chapter, in addition to those already identified, was then to establish the impacts of different grassland fertilisers on water quality and soil pH. The quality of both soil water and run-off water would be assessed by measurement of DOC, nitrate leaching, chloride leaching, phosphate leaching, pH and electrical conductivity. Although not specifically indicated by the NT as a water quality component of concern, measurement of DOC was considered of vital importance, as DOC concentration can cause large variations in water colour, impacting directly on stream water life and water treatment costs (Worrall et al., 2007). In addition to revealing the impacts on water quality it was also hoped that measurements of DOC would further clarify the affects of grassland fertiliser application on the carbon losses/gains from these land-management practices.

4.1.2 Fertiliser application and its impacts on grassland SOC and biomass: A review of the literature

Research into the impacts of fertiliser application on %SOC under grasslands is very sparse. Although the impacts of fertiliser application on agricultural land, and its effect on %SOC has been investigated, the majority of this research has been undertaken with particular reference to arable land (Roupp, 2001; Gong et al., 2009; Triberti et al., 2008; Purakayastha et al., 2008), with a distinct lack of research relative to managed grasslands.

In relation to the impacts on arable cropping systems, Purakayastha et al. (2008) found that SOC increased following NPK fertilisation. The authors attributed this to an increase in plant growth and a subsequent greater return of crop residues to the soil to then be converted to, and stored as SOC. As this study was specific to arable land it is inappropriate to assume that SOC will behave similarly under pasture treated with the same fertilisers, as land-management techniques between the two land-uses vary considerably, with soil tillage and disturbance under arable land allowing a much greater amount of crop residue to be incorporated into the soil than under grasslands. Uncertainties regarding the impacts of fertilisation on arable %SOCs are already high, and therefore not only would it be inaccurate to assume the same impacts on grassland, due to different soil conditions and interactions, but the already high uncertainty regarding this land-use alone means that no firm conclusions can be drawn. This uncertainty is indicated in a study by Khan et al (2007), where in contrast to the findings of Purakayastha et al (2008) mineral fertilisation of arable systems did not increase %SOC. Further major uncertainties relating to the impacts of arable fertilisation on %SOC are discussed in Chapter 3.

The uncertainties relating to the impacts of mineral fertiliser application on %SOC outlined so far are not specific to any type, or form, of mineral fertiliser. With the aims of this chapter being to assess whether phosphate based fertiliser can increase %SOC, literature searches were undertaken to look specifically at the impacts of this type of fertiliser. As with mineral fertilisation in general, the specific impacts on %SOC of phosphate fertilisers was researched in much greater depth for arable land-use. A study by Halvorson and Reule (1999) found significant increases in SOC under plots treated with phosphorus fertiliser, and as no surface residue had been incorporated into the soil they attributed these changes to stimulated root growth. The arguments above relating to soil disturbance and residue incorporation can not be used in this instance to argue that the impact on %SOC will be different under grasslands, as the results appear to have been caused by enhanced root growth, however until more specific results are presented for grasslands, phosphate fertilisers can not be assumed to enhance %SOC beneath this land-use.

All of the literature discussed so far has referred to the measurement and analysis of the direct impacts of fertilisation on SOC, however, the difficulty in measuring the impacts of land-management change on %SOC over a short time period (Williams et al., 2008; Piao et al., 2001) means that several other authors have approached this task by monitoring and comparing gaseous carbon fluxes. The difficulty in directly measuring a change in %SOC stems not only from a large spatial variability in %SOC, but also from the large length of time taken for %SOC to reach a new equilibrium following land-use or land-management change (Heim et al., 2009). A greater amount of research into the impact of fertilisation on gaseous carbon fluxes than that looking specifically at %SOC suggests that this research could be used to help resolve the aims of this chapter. However, as with the impacts on %SOC, the majority of this work is again specific to arable land. Kaboneka et al (1997) studied the effects of nitrogen and phosphorus fertilisation on the decomposition of wheat straw, and found significant increases in decomposition rate with increasing nitrogen and phosphorus application. These increases in decomposition and potential losses of CO_2 to the atmosphere are however thought likely to increase nutrient availability for plant uptake, and hence possibly increase carbon sequestration due to enhanced crop growth. As indicated, this research is specific to arable land, and as with the impacts on %SOC the same assumptions can not be made as regards the impacts of grassland fertilisation on grass decomposition.

In contrast to research into the impacts on %SOC and gaseous carbon fluxes, there is a greater amount of literature specific to the impacts of fertiliser on grassland productivity and the yield of both above and below-ground biomass. Research has not only assessed the impact of fertilised versus non-fertilsed grasslands, but has also assessed the impact of a variety of fertiliser types and amounts, and research specific to the impact of phosphate fertiliser on grass below-ground root productivity has been undertaken. An adequate phosphorus supply is said to promote root growth (Havlin et al., 2005), supporting the results of Halvorson and Reule (1999) discussed above in relation to arable land. These results suggest that in addition to an increase in biomass carbon sequestration, phosphate based fertilisers could also promote SOC accumulation, due to increased root growth and root exudates into the soil. An Increase in root productivity and humus formation with fertiliser application is described by Jollans (1985), and soils under fertilised plots at Rothamsted are reported to have higher SOM than un-fertilised plots due to increased soil inputs from plant roots (Rowell, 1994). Other research specific to arable land reports that balanced fertilisers increase plant biomass (Hati et al., 2008) and are likely to result in increased organic material returns to the soil, however as argued above the same cannot be assumed of grassland systems until more research is undertaken. It is also indicated in the literature that an increase in biomass yield may not correspond with an increase in SOC, and that the results regarding the impact on one carbon pool cannot be assumed to apply to the other. Although in a study by Halvorson and Reule (1999) biomass and crop residue increases were accompanied by an increase in SOC, in a later study by Halvorson et al (2002) crop residue returns increased, but SOC did not.

Although not indicative of the net effect on %SOC there has also been some research into the effects of mineral fertilisers on soil microbial populations and microbial activity (Lima et al., 1996) which could help to resolve some of the aims of this chapter. Lima et al (1996) found increases in microbial populations with increased phosphorus fertiliser addition, relative to a control of no treatment; however application rates of greater than 100Kg/ha resulted in a decline in the microbial population. Despite this, they state that phosphorus is an essential requirement of microbial population growth, a factor which could indicate SOC accumulation. In contrast, the application of nitrogen fertiliser has been found to cause acidification (McAndrew and Malhi, 1992), and as a consequence, a decline in microbial biomass. If the microbial population is influenced by mineral fertiliser addition then it is also likely to influence total SOC, but research into the impacts on microbial activity does not reveal the net affects on SOC, and clearly more research is needed.

As a clear aim of this chapter is to identify the impacts of grassland fertiliser application on DOC losses from soils (Section 4.1.2), literature searches were undertaken to identify any previous results and outcomes. Literature regarding the impacts of land management on mineral soils and its effects on DOC is, however, limited, and literature specific to grassland fertilisation on mineral soils and its impacts on DOC is very sparse. This lack of data is noted by McTiernan et al. (2001), who reveal that the focus of DOC research is concentrated on moorlands and forests, and that the lack of information relating to managed grasslands calls for much greater research. In an attempt to establish some of the impacts, these authors compared DOC fluxes from unfertilised plots to those treated with ammonium nitrate fertiliser on both drained and un-drained soils. When comparing the whole dataset, no statistically significant differences in DOC flux was observed, however on the un-drained plots the flux of DOC was significantly greater from plots treated with nitrogen. McTiernan et al (2001) attribute these increases to increased dry matter production under nitrogen fertilised plots and hence greater leaf and root death, contributing to greater SOM. This increased SOM would result in a greater source for DOC production, and the authors argue that the differences are significant beneath un-drained plots due to the anaerobic conditions allowing SOM to accumulate.

The impacts of grassland fertilisation on soil pH are also studied in this chapter, and it is possible that the liming ability of basic slag and possible acidifying affect of nitrogen (Hati et al., 2008) could result in a different soil pH response to the various fertiliser treatments. Any impacts on soil pH could then have an indirect affect on %SOC, as according to Hati et al (2008) lime application can result in major increases in crop yields which subsequently cause increased root and crop residue decay, increasing carbon returns to the soil, and possible increases in %SOC. These authors found SOC to have a positive relationship with soil pH. In contrast to fertilisers which increase soil pH, Sharma and Subehia (2003) found soil pH and crop yields to decrease under plots fertilised with nitrogen.

4.2 Materials and Methods

4.2.1 Study site

The trial was undertaken on a field on the NT's Wallington Estate in north east England (Section 1.2). The field was chosen due to its uniform land-use, topography and small variation in soil series and altitude, allowing a largely controlled experiment to be undertaken with few uncontrolled variables (Figure 4.1 a-c).

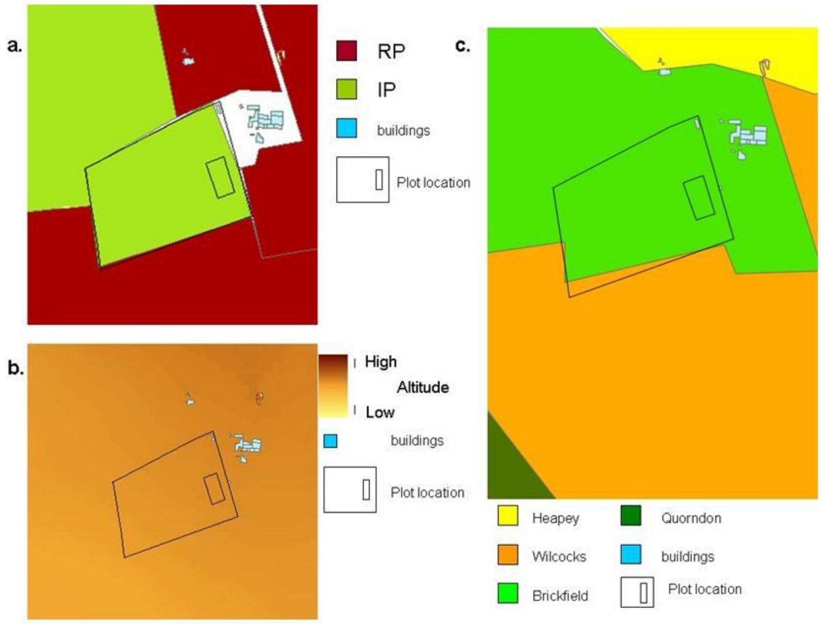

Figure 4.1 a. Plot location under a constant land-use; b: plot location with little variation in altitude; c: plot location under a constant soil series;, Where RP: rough pasture; IP: improved pasture; Heapey, Wilcocks, Brickfield and Quorndon refer to soil series

The specific field chosen was also one that had no history of basic slag application. Choosing a field previously fertilised with basic slag would mean that SOC would not adjust to a new equilibrium, as no land-management change would have taken place, and the aims of this chapter would not be met. The field was grazed by sheep throughout the trial period, other than during one of the winter months, and in May and June when livestock was removed to allow the grass to be grown for silage. The presence of grazing livestock was a result of the trial being undertaken on a working farm, and the fact that it was indicative of a real-world situation. The fact that grazing is an uncontrolled variable in this trial means it must be taken into consideration in the analysis of all results, as it is possible, although not known, that sheep may have favoured certain plots more than others.

4.2.2 Trial design and set-up

Fertiliser treatments were applied to a total of 10 plots, each measuring 5m x 5m on 30th June 2009. Plots were marked out using pegs and string, and were separated by 2m unfertilised strips to minimise any contamination which could occur during fertiliser spreading and application, and as a result of wind and rain throughout the trial period. As explained in Section 4.2.1 a field with uniform vegetation, soil series, slope and altitude was chosen in an attempt to ensure that all plots were subjected to the same conditions. The 10 plot design consisted of four fertiliser treatments, each in replicate, and two plots where no fertiliser was applied, to act as a control. The plots were laid out in a completely randomised design..

The four fertiliser treatments were:

1. NPK: in the ratio 20:10:10, at an application rate of 92 kg/ha (0.23 kg/plot)

2. Nitrogen fertiliser: 20% N, at an application rate of 62 kg/ha (0.16 kg/plot)

3. Basic slag at an application rate of 2470 kg/ha (6.18 kg/plot)

4. A combination of basic slag and nitrogen fertiliser applied at the same rates as in 2 and 3.

The fertiliser treatments chosen were based on the findings of the research in Chapter 3, which show greater %SOC's under pastures fertilised with basic slag than those fertilised with NPK or nitrogen. The rates of application were chosen based on the results of the land-management questionnaires in Chapter 3, and the information provided by farmers relating to their typical rates of rough pasture fertilisation (Section 3.4.1.2).

To allow measurement of NER, NEE and PP as described in Section 4.2.4, three soil respiration collars were installed into each plot. Each collar was constructed from a section of six inch diameter drain pipe of 10 cm length. The respiration collars were inserted in random locations within each plot to a depth of approximately 5 cm, leaving 5 cm protruding from the surface (Figure 4.2 a and b). These collars were installed on the day of fertiliser spreading and, so as to minimise effect of disturbance upon subsequent measurements, were left in place for two weeks before any measurements were taken. The collars provided a surface onto which the gas chamber could be placed when making NER, NEE and PP measurements, as described in Section 4.2.4. Positioning the chamber onto the collar meant that an air-tight seal could be achieved.

Figure 4.2 a. Respiration collar b. Randomly located collars within each plot

4.2.3 Pre and post-trial %SOCs

Although it was thought unlikely that a difference in %SOC would be evident between plots treated with different fertilisers after only one year, due to the large natural variability in SOC (Williams et al., 2008; Piao et al., 2001), and the length of time required for %SOC to adjust to a new equilibrium following land-use change (Heim et al., 2009), it was still considered important to investigate whether any differences were apparent. The inclusion of two control plots in the trial design (where no fertiliser was applied) meant that the %SOC of each fertilised plot could be compared to the control at the end of the trial to identify any increases or decreases in %SOC with fertiliser application. The large spatial variability and natural variation in %SOC often found beneath small areas of land meant that in addition to comparing %SOC from the control and fertilised plots at the end of the trial, it was also decided to measure %SOC beneath each plot prior to fertiliser application, and to compare these measurements to the %SOC beneath each plot at the end of the trial. The large spatial variation often found in %SOC (Saby et al., 2008; Tolbert et al., 2002; Chapter 2) meant that a high sampling density was required to obtain an accurate mean %SOC for each plot. A total of six soil samples were taken from a depth of 20 cm (the depth appropriate to detect the effects of land-use change: see Section 2.2.2) beneath each plot in June 2009, prior to any fertiliser application. These samples were taken from six random locations beneath each plot using a soil auger and collecting a sample from the 18-22 cm layer. The same procedures and sampling density were used to collect the samples one year after fertiliser application. Following soil collection at both the beginning and end of the trial all samples were transported immediately back to the laboratory where they were dried overnight at 105°C, and treated and analysed for %SOC following the methodology outlined in Section 2.2.2.1. This resulted in the total analysis of 120 soil samples (60 pre trial and 60 post trial), with 12 from beneath each fertiliser treatment on both occasions, allowing statistical analysis of the difference in %SOC between treatments to be undertaken.

4.2.4 Carbon flux measurements

4.2.4.1 NER

Measurements of NER were made from the ground surface on a fortnightly basis from August 2009 to July 2010, and include respiration from the above and below-ground vegetation, roots and soil. These measurements represent the total amount of carbon released from the ground surface in g C/m^2/hr. The first measurements were made on 6th August 2009 between 1000 and 1600 hours, and every fortnight following to gain an accurate estimate of the seasonal variation in flux over an annual period. As this measurement represents the flux of carbon from the land to the atmosphere it is always a positive number, the more positive the number the greater the release of carbon to the atmosphere.

NER was measured using a portable IRGA (PP systems EGM-4, Hitchin, UK) as shown in Figure 4.3.

Figure 4.3 IRGA used to measure CO_2 flux from/to the atmosphere

A clear acrylic chamber attached to the IRGA was fitted onto the permanently installed collars and a tight seal was ensured between the chamber and the collar. The acrylic chamber was covered with an opaque cover to stop any photosynthesis (and hence carbon uptake from the atmosphere) from taking place. The chamber was installed with a small fan to ensure sufficient mixing of air and to allow the chamber to be purged of gas in between flux measurements so that the air in the chamber could re-equilibrate with the atmosphere. The air within the chamber was pumped from the chamber into a sample cell within the IRGA and back again. The amount of CO_2 within the sample cell was calculated by the IRGA based on the amount of infra-red light emitted and received. The CO_2 concentration (parts per million by volume (ppmv)) was measured and recorded by the IRGA every 4 seconds for a total of 124 seconds. This information was then used to calculate the carbon flux from the ground surface to the atmosphere. The flux value was calculated based on the ideal gas law:

$$PV = nRT$$ ***Equation 4.1***

Where P = pressure (atm), V = system volume (l), n = number of moles, R = universal gas constant (l atm/mol/K) and T = temperature (K).

Equation 4.2 was then used to calculate the weight of CO_2 within the chamber:

$$G = 1 \times 10^6 \, [CO_2] V \left(\frac{p}{nRT} \right) M_r$$ ***Equation 4.2***

Where G = weight of gas (g), $[CO_2]$ = concentration of CO_2 (ppmv), V = volume (l), P = pressure (atm), n = number of moles, R = universal gas constant (l atm/mol/K), T = temperature (K), M_r = relative atomic mass (g/mol).

Equation 4.3 was then used to calculate the flux in the chamber:

$$F = \left(\frac{C_1 - C_0}{Time} \right) / SA$$ ***Equation 4.3***

Where F = flux (CO_2/g/m^2/hr), C_1 = CO_2 weight within the chamber at time 1 (g), C_0 = CO_2 weight within the chamber at time 0 (g), time = time between 1 and 0 (h), SA = surface area (m^2).

As this is the flux of CO_2 in the chamber it was converted to the flux of carbon by dividing the answer by 3.66.

4.2.4.2 NEE

Measurements of NEE were also made from the ground surface every fortnight from August 2009 to July 2010. These measurements represent the difference between the total amount of carbon released from the ground surface and that taken up from the atmosphere in PP. Measurements of NEE began on 16th August 2009, and were taken between 1000 and 1600 hours. Measurement of NEE was undertaken in a similar way to NER, using a portable IRGA and soil chamber; however the clear acrylic chamber was left uncovered. The use of a clear chamber meant that both respiration and photosynthesis could occur, thus providing a carbon flux value from or to the atmosphere depending on the balance between these two fluxes. The measurements refer to g $C/m^2/hr$. By convention a net flux of CO_2 to the atmosphere is given a positive value.

4.2.4.3 PP

As with NER and NEE, calculations of PP were made every fortnight from August 2009 to July 2010. These calculations represent the total amount of carbon taken in from the atmosphere by the vegetation growing on the soil surface, and the below-ground vegetation and roots. PP is represented as a negative number (the more negative the number the greater the uptake of carbon from the atmosphere). The PP values refer to the uptake of carbon from the atmosphere in g $C/m^2/hr$ and were made using measurements taken between 1000 and 1600 hours on every visit. The PP values were not measured directly, but were calculated from the direct measurements of NEE and NER, with PP being the difference in the two measurements. PP was calculated using Equation 4.4.

$$PP = NER - NEE$$ **Equation 4.4**

4.2.4.4 Surface-air temperature and PAR

Measurements of surface-air temperature at the time of each IRGA reading were required, as indicated in Equation 4.1 and Equation 4.2. A temperature sensor was therefore installed within the chamber, meaning that surface-air temperature was recorded along with every reading of CO_2 taken over each 124 second IRGA reading. As a variable over which there could be no experimental control, it was also important to measure and record photosynthetically active radiation (PAR) as this can vary greatly over very short timescales due to variations in weather conditions and cloud cover. Measurement of PAR was undertaken along with every CO_2 reading by a PAR sensor located within the chamber.

4.2.5 Biomass carbon stocks

Although fortnightly IRGA PP measurements will reveal the mean PP under different fertiliser treatments for each period of measurement, it was also thought beneficial to sample the total above-ground biomass accumulation from each plot following one year of fertiliser application. These measurements would show how total accumulation varies with fertiliser treatment, and be used in support of the IRGA readings. Total above-ground biomass from the area within each soil respiration collar was harvested from the site in July 2010 following the final IRGA readings. The randomly located respiration collars provided a total of six biomass carbon accumulation figures for each fertiliser treatment. The biomass was harvested by cutting all above-ground vegetation down to the soil surface, but removing no roots in the process. This was then placed in sample bags, labelled, and transported back to the laboratory where it was dried in an oven overnight at 70°C to remove all moisture content. The samples of biomass were weighed and their weight recorded. As the area of each respiration collar was known these values of biomass were converted to values of mass of carbon/ha, with the biomass carbon content assumed to equal 50% (Singh and Lodhyial, 2009).

4.2.6 Soil water chemistry and water table depth

On the day of fertiliser application a 100 cm long dipwell was inserted at least 90 cm into the ground of the centre of each plot (Figure 4.4). This was done using an auger to remove a soil core of > 90 cm depth, followed by insertion of the dipwell into the hole left by removal of the core. Dipwells were constructed from 3 cm diameter drain pipe of 100 cm length, with two holes drilled at every 10 cm down the length of the pipe, and the ends of the pipe left open. These holes allowed groundwater to enter the dipwell and water table depth to be measured on each fortnightly visit to the site, by inserting an electrical conductivity probe into the dipwell and recording the depth at which water contact was made.

Figure 4.4 Dip-well inserted > 90cm into ground

In addition to water table depth measurement, a sample of soilwater was extracted from the dipwells for water chemistry analysis on each visit to the site. These samples were transported back to the laboratory and either analysed immediately or refrigerated at < 4 °C for future analysis. Each sample was filtered to < 0.45 µm with cellulose acetate syringe filters, and analysed for pH and electrical conductivity using electrode methods (pH meter, HI-9025; conductivity meter, HI-9033, Hanna instruments). This was done on a groundwater sample from each plot in order to establish whether the type or presence of fertiliser had any impact on the pH and electrical conductivity of groundwater, and to meet the aims established in Section 4.2.1. The DOC concentration of the sampled waters was measured using the colorimetric method of Bartlett and Ross (1988). In this method Mn (III) is reduced by the OC within the water sample in the presence of sulphuric acid, leading to a loss of colour. Measurement of this loss of colour then enables the amount of OC within the water sample to be determined. Absorbance changes were measured spectrophotometrically at 495 nm. Determination of the DOC concentration of the water samples is then undertaken by reference to a calibration graph produced from samples of known concentration and their absorbance at 495 nm. Measurement of DOC concentration was undertaken as indicated in Section 4.2.1, to establish the impacts of fertilisation on not only environmental and water quality issues, but also to measure any losses of carbon not detected in studies looking only at gaseous fluxes.

All samples were also analysed for their anion concentrations by ion chromatography (Metrohm, Compact IC 761). In this method the components for analysis travel along in a fluid phase (consisting of sodium carbonate and sodium hydrogen carbonate) over a stationary phase with a large surface area. The different anions within the water sample pass through the column at different stages depending on their affinity with the fluid phase. As each anion passes through the detector its change in concentration is measured, and a chromatogram produced by the machine reveals the concentration of the anions as they exit the column. A number of samples (standards) containing known amounts of each anion, and 5 blanks were made up and run prior to any analysis, and following the analysis of every 20 samples. The following standards were run: 1.25ppm, 2.5ppm, 5ppm, 10ppm, 25ppm, 50ppm and 100ppm, and as these samples were of a known concentration their intensity was plotted and a calibration graph produced. It was ensured that the range of concentrations covered by the water samples was also covered by the standards. Identification of each anion was done by comparison with the retention time of each anion in the standards, and the concentration of each was then determined by comparison of the peak size with the peaks of known concentration and the calibration graphs produced from the standards. Although each of the anions fluoride, chloride, bromide, nitrate, phosphate and sulphate were analysed by this method particular reference was made to nitrate and phosphate concentrations, as these were considered the most relevant in relation to fertiliser losses to the environment.

4.2.7 Run-off water chemistry

Following fertiliser application three run-off traps were installed into each plot to collect water draining from the ground surface, and to assess the quantities of fertiliser retained, and any affects of fertiliser application on DOC, run-off water pH and electrical conductivity. Run-off traps were constructed from a 20 cm length of 3 cm diameter drain pipe with four holes drilled at right angles approximately 5cm from the top of the pipe. A rubber bung inserted into both the top and bottom of the pipe ensured that once inserted into the ground no water could enter the pipe from soil water, or directly as rainfall, and that all water collected in the pipe had accumulated directly from run-off. A soil auger was used to remove a 15 cm long core from the ground surface at three random locations within each plot. The cores removed were discarded and the run-off traps were inserted into the holes left by core removal. The holes in the sides of the run-off traps were positioned level with the ground surface to allow surface run-off to accumulate (Figure 4.5).

Figure 4.5 Run-off traps

Each run-off trap was inspected on fortnightly visits to the field site and if sufficient water had collected this was pumped out of the trap into a container and transported back to the laboratory for water chemistry analysis. All water samples were filtered to < 0.45 µm and analysed for pH, electrical conductivity, DOC concentration and anion concentrations as in Section 4.2.5.

4.2.8 Pre and post trial soil pH

Soil pH was measured on every soil sample collected and analysed for %SOC in Section 4.2.3. This resulted in a total of 120 soil pH measurements, 60 pre trial and 60 post trial, consisting of 12 samples from each of the fertiliser treatments on both occasions. As with the samples for %SOC analysis, soil pH from beneath the control plots at the end of the trial were compared to soil pH from the fertilised plots at the end of the trial. The same issues regarding large spatial variations in %SOC were however also likely to apply to the natural variation in soil pH. It was for this reason that soil pH measurement was undertaken on all soil samples prior to fertiliser application, so that in addition to comparing post trial pH from fertilised plots to control plots, a comparison of soil pH pre and post fertiliser application could also be made if soil pH varied significantly across the field site.

4.2.9 Statistical analysis

All analysis of %SOC, NER, NEE, PP, soil pH and soil water chemistry data was performed using Minitab 14 statistical analysis software. Comparison of %SOC between plots prior to the start of the trial was done by one-way ANOVA, as the nature of the study site meant that all other variables were controlled (see Section 4.2.1). The same statistical analysis was used to compare %SOC between land-uses at the end of the trial, and to compare the difference in %SOC pre and post trial. Analysis of soil pH between fertilised treatments was undertaken in the same way to that of %SOC. Analysis of NER, NEE and PP between land-uses was undertaken using ANOVA, with ANCOVA then used to establish if these land-use controls remained significant with uncontrolled experimental variables held constant. Statistical analysis by ANCOVA was chosen as it is a method specifically used on categorical variables, allowing the variability among group means to be compared with the variability between group means. ANCOVA allows the main effect of the factor/categorical variable to be identified by controlling for the effects of other continuous variables/covariates. This method of analysis removes the effects of variables which modify the relationship between the independent and dependent variables, producing an adjusted mean (an estimate of the true mean if these variables were controlled). The r^2 values generated by ANCOVA represent the between *group sum of squares* divided by the *total sum of squares,* with a large r^2 value thus indicating that a large fraction of the variation in the independent variable can be explained by the categorical variable/treatment. The r^2 value represents the proportion of the total variation explained by the difference in the means. ANCOVA allowed any significant difference between the factor under study to be identified when other factors and covariates were controlled statistically in the analysis. The factors investigated in this study as controls on NER, NEE and PP were fertiliser treatment and month of measurement. A variation in surface-air temperature with measurement meant that surface-air temperature was entered as a covariate in all analyses. Water table depth and PAR were also considered as covariates in all analyses, and PP was considered as a covariate in the analysis of NER. Analysis of DOC in both soil water and run-off water, and its variation with fertiliser treatment was undertaken using GLMs and ANOVA, with fertiliser treatment and month of year entered as factors in the analysis. ANCOVA was then undertaken to establish whether fertiliser treatment remained significant with other variables held constant. In this analysis water table depth, soil/run-off water pH, soil/run-off water electrical conductivity and the concentration of each of the analysed soil/run-off water anions were included as covariates. Analysis of the variation in nitrate and phosphate concentration with fertiliser treatment in both soil water and run-off water was undertaken using GLMs and ANOVA, with fertiliser treatment and month of measurement entered as factors in the analysis. ANCOVA was then undertaken on these samples, with water table depth, run-off/soil water pH, and run-off/soil water nitrate/phosphate concentration included in the analysis as covariates. Before any analysis was undertaken it was ensured that all data were normally distributed. Any significant difference identified by ANCOVA or ANOVA was then post-hoc tested using the Tukey test, to identify between which factors significant differences in %SOC, NER, NEE, PP, soil and run-off water DOC, soil and run-off water nitrate concentration, and soil and run-off water phosphate concentration, soil and run-off water pH and soil pH occurred. Results were considered to be statistically significant when $p<0.05$. Main effects plots were generated to display the adjusted means of the variable under study for each of the independent factors. The main effects plots show the mean for each factor with the effects of other variables removed. Each point on the main effects plots is the mean of all measurements taken over the trial period for that factor, with the horizontal line representing the overall mean of the entire dataset.

4.3 Results

4.3.1 Change in %SOC with fertilisation

Collection of soils prior to fertilisation resulted in 54 samples for analysis of %SOC. This consisted of 12 from the control plots and those to be fertilised with basic slag, 8 from the plots fertilised with a combination of basic slag and NPK, and 11 from plots fertilised with nitrogen and those with NPK. The inconsistency in sample number between plots resulting from sample loss during transportation and storage. Initial %SOC varied across the field from a low of 3.40% to a high of 4.23%. This is likely to be the result of a naturally large spatial variation in %SOC. The lowest mean %SOC of 3.40% was found under plot 4, which was to be fertilised with nitrogen, and the highest mean of 4.23% under plot 2, which was to be fertilised with a combination of basic slag and nitrogen. Despite the variation in mean %SOC between plots there were no statistically significant differences.

Collection of soils one year after fertiliser application resulted in 60 samples for analysis of %SOC. This consisted of 12 samples from each of the treatments, with %SOC varying across the field from a low of 2.37% to a high of 3.57%. The lowest mean %SOC of 2.37% was found under plot 5, a plot which had been fertilised with both basic slag and NPK, and the highest mean %SOC of 3.57% under plot 7 which had been fertilised with nitrogen (Figure 4.6).

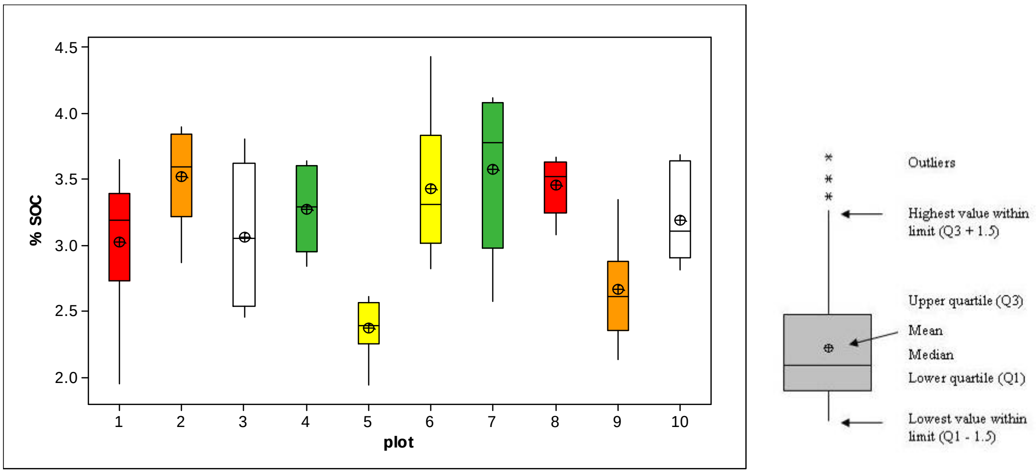

Figure 4.6 Variation in %SOC beneath and between plots one year after fertilisation, where red: basic slag; orange: NPK; white: control; green: nitrogen; yellow: basic slag and nitrogen

One-way ANOVA revealed a statistically significant difference in %SOC with plot; however when post trial %SOC was analysed by fertiliser treatment rather than by plot there was found to be no significant difference (Figure 4.7).

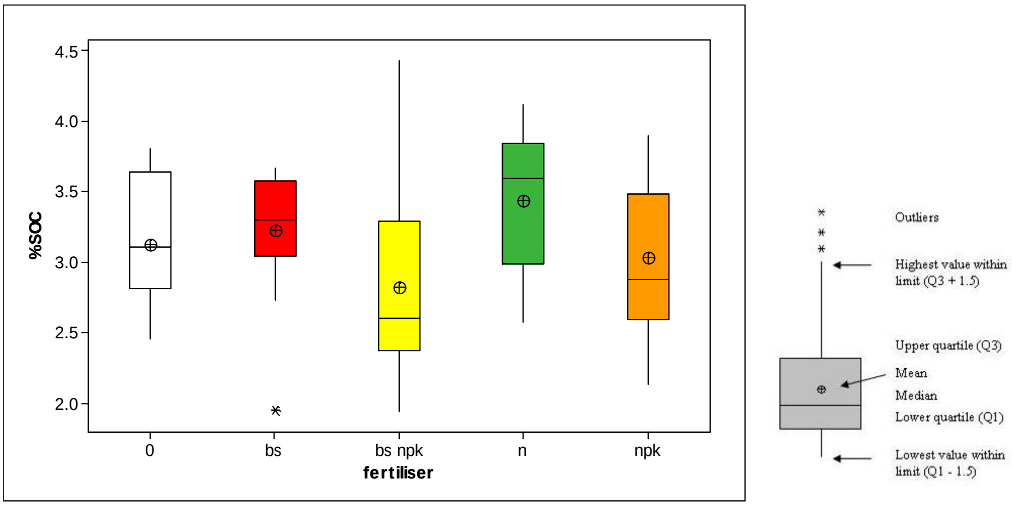

Figure 4.7 Post trial variations in %SOC beneath and between fertiliser treatments

Although %SOC did not vary across the field site prior to fertilisation, it was decided that comparison of before and after treatment %SOC should still be made. This revealed a significantly lower %SOC under all treatments other than the nitrogen treatment one year after fertilisation. %SOC under the nitrogen treatment was maintained (Figure 9.7).

All data relating to %SOC before and after fertiliser application can be found in Appendix 6 under the heading '%SOC data'.

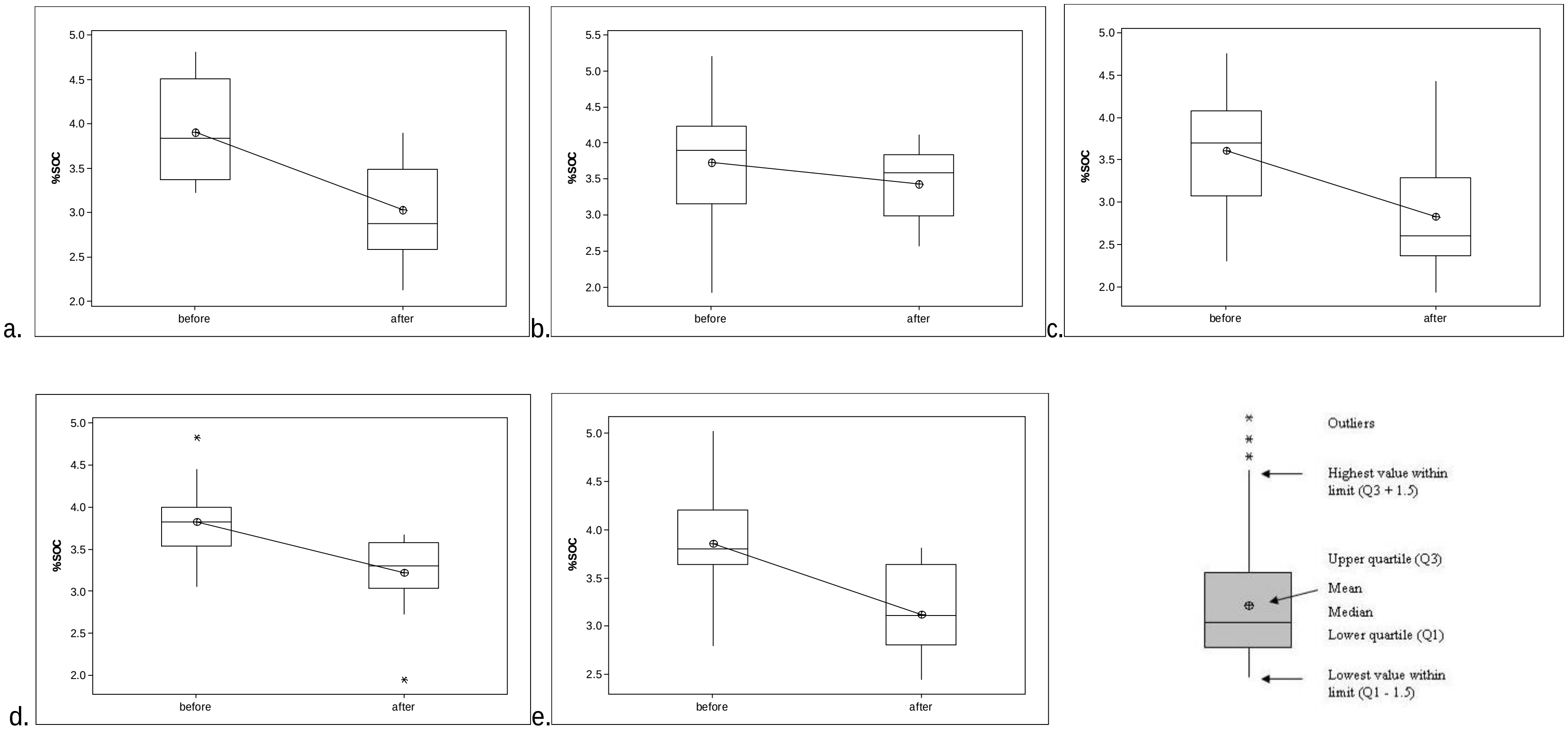

Figure 4.8 A change in %SOC following one year of fertilisation under plots treated with a. NPK; b. nitrogen; c. basic slag + NPK; d. basic slag; e. untreated (control)

4.3.2 Gaseous carbon fluxes

All gaseous carbon flux raw data and covariate measurements can be found in Appendix 6 under the heading 'CO_2 flux data'.

4.3.2.1 NER

Measurement of NER from each collar within each plot every fortnight resulted in a total of 595 readings, with a mean NER from all treatments over the measurement period of 0.25g C/m^2/hr. These measurements included 99 from the control plots, 124 from the plots fertilised with basic slag, 124 from the plots fertilised with both basic slag and NPK, 121 from the plots fertilised with nitrogen, and 127 from the plots fertilised with NPK. The variation in measurement number with treatment was the result of equipment failure and anomalous flux readings in the winter months, and the inability to locate one of the collars on the control plot due to grassland growth. Although NER was not normally distributed (due to a large number of very small fluxes in the winter months) log transformation did not improve the distribution; therefore NER was not log-transformed in any data analysis. A two-way ANOVA analysing the affect of month of measurement and fertiliser treatment on the variability in NER revealed that variability in fertiliser treatment had no significant effect. Inclusion of the covariates: water table depth, PP, and surface-air temperature in an ANCOVA, revealed that all of these covariates could explain some of the variation in NER, but fertiliser treatment remained insignificant. Together with month of measurement they could explain 74.05% of the variation in NER. The insignificant affect of fertiliser treatment on grassland NER is demonstrated in Figure 4.9, and the seasonal cycle in NER can also be observed, with the greatest fluxes in the summer months, and the smallest fluxes in the winter months.

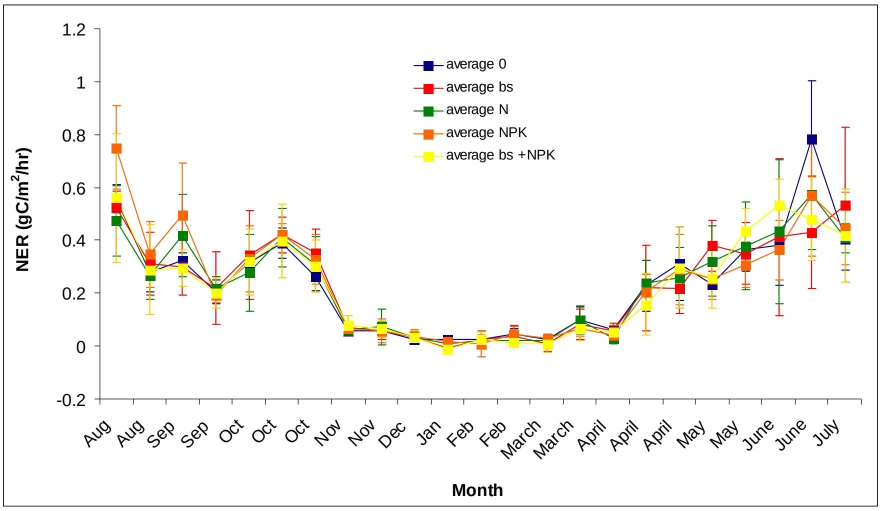

Figure 4.9 Seasonal variation in NER and the insignificant affect of fertiliser, where 0: control; bs: basic slag; N: nitrogen; NPK: NPK; bs + NPK: basic slag and NPK; where symbols represent the mean and error bars represent the standard deviation

4.3.2.2 NEE

Fortnightly measurements of NEE resulted in a total of 589 readings, with a mean NEE from all treatments over the entire measurement period of -0.06 g C/m^2/hr. These measurements included 101 from the control plots, 120 from the plots fertilised with basic slag, 123 from the plots fertilised with both basic slag and NPK, 119 from the plots fertilised with nitrogen, and 126 from the plots fertilised with NPK. The variation in measurement number with treatment can be explained by the issues described in Section 4.3.2.1.

Two-way ANOVA analysing the affects of month of measurement and fertiliser treatment on NEE revealed that both factors had a significant impact on NEE. Post-hoc analysis of the data indicated that the plots treated with nitrogen fertiliser had a significantly greater (more negative) NEE than the un-fertilised plots. The interaction between month of measurement and fertiliser treatment was not significant, indicating that the impact of fertiliser treatment did not vary depending on the time of year. The covariate PAR was log transformed as it was not normally distributed, and the covariates water table depth, surface-air temperature and PAR were included in the analysis. Although surface-air temperature and water table depth were unable to explain any of the variation in NEE, PAR was found to have a significant affect, and its inclusion in the analysis resulted in fertiliser treatment becoming insignificant. This suggests that PAR varied with fertiliser treatment, and that variation in fertiliser treatment itself is not responsible for any of the variation in NEE. The suggestion of a systematic variation in PAR with treatment is however unlikely, indicating that fertiliser treatment may still be responsible for some of the variation in NEE. The variation in month of measurement and fertiliser treatment could together explain only 18.66% of the variation in NEE. The variation in NEE with fertiliser treatment and season can be observed in Figure 4.10. The main effects plot is displayed in Figure 4.11, and shows the adjusted mean NEE values for each fertiliser treatment over the trial period, when controlled for the other variables.

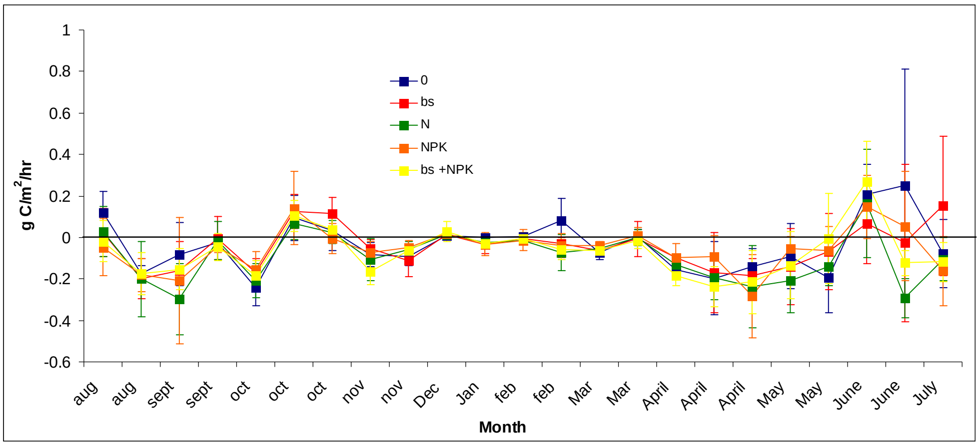

Figure 4.10 The variation in NEE with fertiliser and season, where 0: control; bs: basic slag; N: nitrogen; NPK: NPK; bs + NPK: basic slag and NPK; where symbols represent the mean and error bars represent the standard deviation

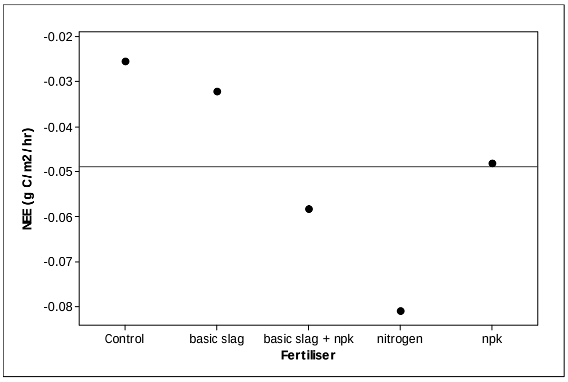

Figure 4.11 Main effects plot showing a significantly greater NEE under nitrogen treated plots than un-fertilised plots over the trial period

4.3.2.3 PP

Calculation of PP using the measurements of NER and NEE described in Sections 4.3.2.1 and 4.3.2.2 resulted in a total of 586 readings, with a mean PP from all treatments over the entire measurement period of -0.31g C/m^2/hr. These measurements included 97 from the control plots, 121 from the plots fertilised with basic slag, 123 from the plots fertilised with both basic slag and NPK, 119 from the plots fertilised with nitrogen, and 126 from the plots fertilised with NPK, with the variation in measurement number with treatment being explained in Section 4.3.2.1.

Statistical analysis via a two-way ANOVA, analysing the effect of fertiliser treatment and month of measurement on PP revealed that variation in fertiliser treatment could not explain the variability in PP, with month of measurement having a significant affect. Inclusion of the covariates water table depth, surface-air temperature and PAR (log transformed) were all found to be significant, and in combination with month of measurement could explain 56.95 % of the variation in PP. The insignificant difference in PP with fertiliser treatment over the trial period can be observed in Figure 4.12.

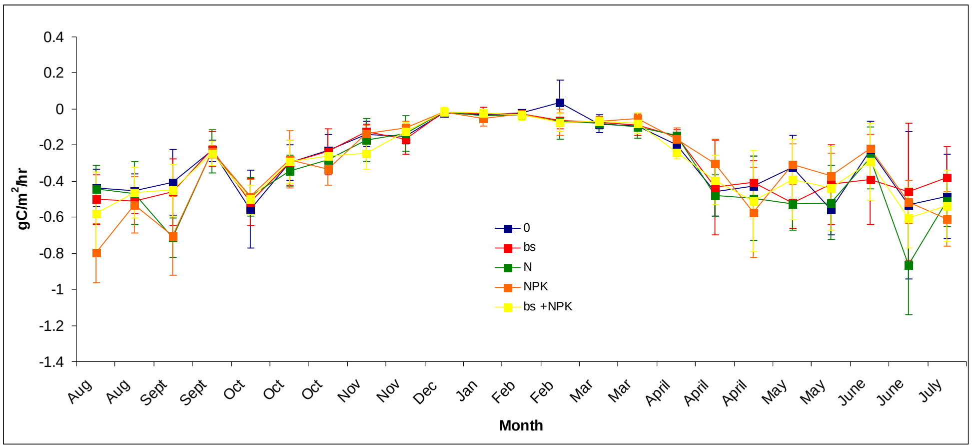

Figure 4.12 The seasonal variation in PP and insignificant affect of fertiliser, where 0: control; bs: basic slag; N: nitrogen; NPK: NPK; bs + NPK: basic slag and NPK; where symbols represent the mean and error bars represent the standard deviation

4.3.3 Above-ground biomass carbon stocks

Measurement of total biomass accumulation for each fertiliser treatment and the control, followed by one-way ANOVA revealed no statistically significant differences in biomass accumulation between treatments. This can be seen in Figure 4.13, and although the mean above-ground accumulation was greatest on plots fertilised with nitrogen, there was a large standard deviation and no significant difference between this and other treatments.

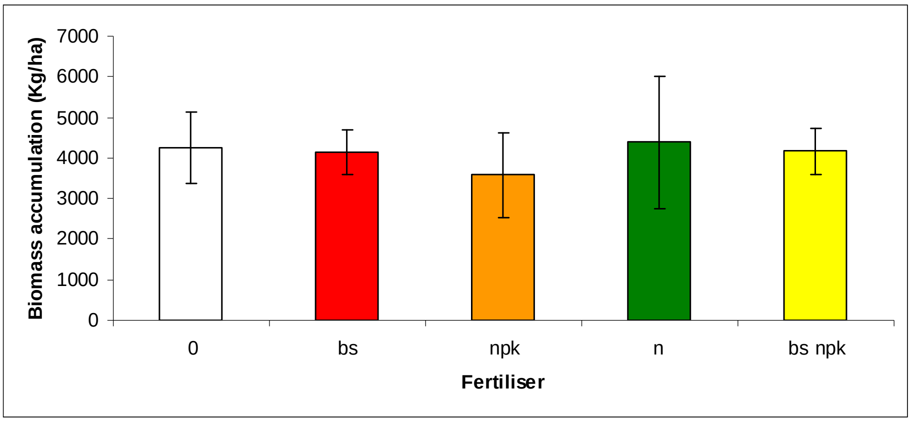

Figure 4.13 The mean and standard deviation of total above-ground biomass accumulation following one year of fertilisation. Error bars indicate one standard deviation from the mean

Biomass accumulation data can be found in Appendix 6 under the heading 'Biomass'.

4.3.4 Soil water chemistry

Soil water sample collection from each dipwell every fortnight from August 2009 to July 2010 resulted in a total of 185 samples for water chemistry analysis. Low water table levels in the drier summer months meant that water samples could not be collected on all visits to the site, and so not all months are represented in the analysis, with no samples collected in August 2009 or July 2010. Analysis of the distribution of soil water DOC revealed that it was not normally distributed, becoming more normally distributed following log transformation. This was also the case with soil water conductivity, soil water chloride concentration, soil water nitrate concentration, and soil water phosphate concentration; therefore these variables were log transformed and their log transformed values used in all analysis.

All raw data relating to soil water chemistry is located in Appendix 6 under the heading 'Soil water chemistry data'.

4.3.4.1 DOC

DOC concentration analysis was undertaken on a total of 185 samples, with 37 samples from each of the five fertiliser treatments. Statistical analysis via two-way ANOVA revealed that date of measurement could explain 87.29% of the variation in DOC, but that fertiliser treatment was insignificant. Inclusion of the covariates soil water conductivity and soil water phosphate concentration in combination with the factor date of measurement explained 92.99% of the variation in DOC, with fertiliser treatment remaining insignificant. All other covariates were unable to explain any more of the variation.

4.3.4.2 pH

In relation to the impact of fertiliser type on soil water pH, analysis was undertaken on a total of 183 samples. 37 samples were analysed from each of the control, basic slag and NPK, and nitrogen treatments, and 36 samples from both the basic slag and NPK treatments. Statistical analysis via two-way ANOVA revealed that both date of sample collection and fertiliser treatment had a significant effect and that variation in these factors could explain 82.23% of the variation in soil water pH. When soil water chloride concentration was included in the analysis alongside the factors date of measurement and fertiliser treatment it was found to be able to explain some of the additional variation in soil water pH. Together these variables could explain 82.85% of the variation in soil water pH, with fertiliser treatment continuing to have a statistically significant effect. None of the other covariates included in the analysis could explain any of the further variation. Post-hoc analysis of the data revealed a significantly lower soil water pH beneath those plots fertilised with NPK than beneath those left un-fertilised, and a significantly lower pH beneath those plots fertilised with nitrogen and NPK respectively than those fertilised with basic slag. The main effects plot showing the variation in pH with treatment, when all other variables are controlled, is displayed in Figure 4.14.

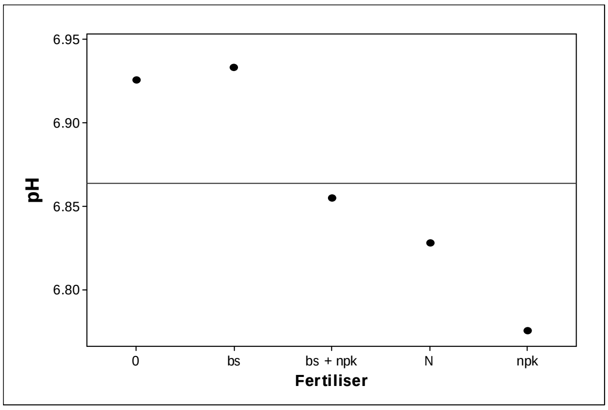

Figure 4.14 Main effects plot showing a significant difference in mean soil water pH over the trial period, where 0: un-fertilised; bs: basic slag; bs + npk: basic slag and NPK; N: nitrogen; npk: NPK

4.3.4.3 Nitrate concentration

Sample collection on a fortnightly basis resulted in the analysis of 134 soil water samples for nitrate concentration. This consisted of 26 samples from the control treatment, 22 from the basic slag treatment, 29 from the basic slag and nitrogen treatment, 25 from the nitrogen treatment and 32 from the plots treated with NPK. Statistical analysis via a two-way ANOVA revealed that both date of measurement and fertiliser treatment could explain 37.97% of the variation in soil water nitrate concentration, both having a significant effect. Inclusion of none of the other covariates in the analysis could explain further any of the variation in nitrate concentration. Post-hoc analysis of the data revealed a significantly lower nitrate concentration leaching from the plots fertilised with basic slag than those fertilised with basic slag and NPK, however there were no other statistically significant differences between any other fertiliser treatments. The main effects plot, indicating the variation in mean nitrate concentration between each fertiliser treatment when controlled for all other variables is displayed in Figure 4.15.

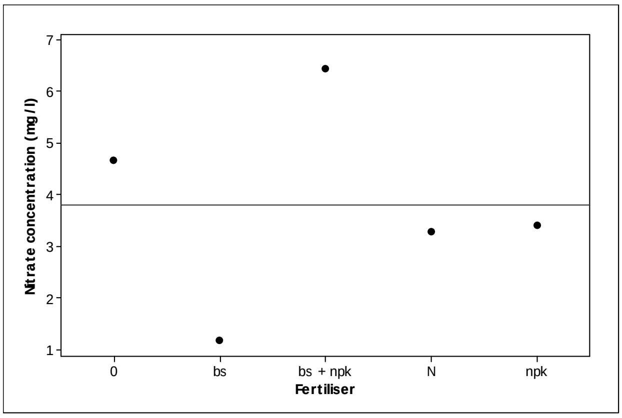

Figure 4.15 Main effects plot showing the mean nitrate concentration over a one year period in soil water from plots receiving different fertilisers, where 0: control; bs: basic slag; bs + npk: basic slag and NPK; N: nitrogen; npk: NPK

4.3.4.4 Chloride concentration

A total of 178 water samples were analysed for chloride concentration, with this analysis consisting of 35 samples from both the control and nitrogen fertilised plots and 36 samples from all other treatments respectively. Two-way ANOVA of the data revealed that both fertiliser treatment and month of measurement had a statistically significant effect on the concentration of chloride in soil water, together explaining 58.78% of the variation. The covariates water table depth and soil water pH could further explain some of the variation in soil water chloride concentration; however the effect of fertiliser treatment remained significant with their inclusion. None of the other covariates could explain any of the further variation in chloride concentration. Together the variation in these covariables and factors could explain 63.02% of the variation in soil water chloride concentration. Post-hoc analysis of the data revealed that the significant difference between fertiliser treatments occurred between the plots treated with a combination of basic slag and NPK and all other treatments, with the basic slag and NPK treated plots having a significantly greater soil water chloride concentration. The main effects plot, displaying the variation in chloride concentration between fertiliser treatments, when controlled for other variables is displayed in Figure 4.16.

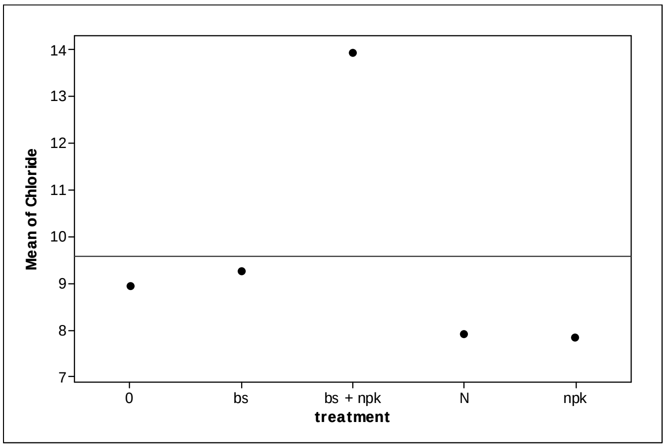

Figure 4.16 Main effects plots showing the mean soil water chloride concentration over a one year period, where 0: control; bs: basic slag; bs + npk: basic slag and NPK; N: nitrogen; npk: NPK

4.3.4.5 Phosphate concentration

Statistical analysis of the data revealed that none of the variation in the phosphate concentration of soil water could be explained by variation in the factors and covariates measured in this study.

4.3.5 Run-off water chemistry

Run-off water was collected from each run-off trap on each fortnightly visit to the site if sufficient water had accumulated to enable analysis. This resulted in a total of 196 run-off water samples. A normality test undertaken on run-off water electrical conductivity, run-off water DOC, run-off water chloride concentration, run-off water nitrate concentration and run-off water phosphate concentration revealed that these datasets were not normally distributed, therefore these datasets were log transformed, and the log transformed data used in all analysis of run-off water chemistry.

All raw data relating to run-off water chemistry can be found in Appendix 6 under the heading 'Run-off water chemistry'.

4.3.5.1 DOC

DOC analysis was undertaken on a total of 182 run-off water samples, consisting of 49 samples from the control plots, 52 from the plots fertilised with basic slag, 32 from the plots treated with a combination of basic slag and NPK, 26 from the nitrogen fertilised plots, and 23 from the plots treated with NPK. The inconsistency in the number of samples analysed from each treatment was the result of inconsistent sample accumulation. Two-way ANOVA revealed that fertiliser treatment had no effect on the concentration of DOC in run-off water, and that date of sample collection alone could explain 57.22% of the variation in DOC, with a median DOC concentration from all plots of 94.93 mg/l (interquartile range: 58.30 – 113.55 mg/l). None of the covariates listed in Section 4.3.5 could further explain any of the variation in DOC concentration.

4.3.5.2 pH

A total of 196 samples were analysed for run-off water pH over the trial period, with 52 from the control plots, 56 from the basic slag fertilised plots, 36 from the plots treated with a combination of basic slag and NPK, 28 from the plots treated with nitrogen and 24 from the plots treated with NPK. As with the samples analysed for DOC, the inconsistency in number of samples analysed per treatment stems from an inconsistency in run-off sample accumulation between plots. Two-way ANOVA assessing the effect of date of measurement and fertiliser treatment on run-off water pH revealed that the application of different types of fertiliser had no significant effect, with 55.23% of the variation in pH being explained by date of measurement alone. Inclusion of run-off water electrical conductivity and chloride concentration in the analysis revealed that variation in these covariates could further explain some of the variation in pH, together with date of measurement explaining 72.42% of the variation in run-off water pH. With the inclusion of the covariates the effect of fertiliser treatment remained in-significant.

4.3.5.3 Nitrate concentration

A total of 153 samples were analysed for nitrate concentration over the trial period, consisting of 38 from the control plots, 41 from the basic slag plots, 32 from the plots treated with a combination of basic slag and NPK, 21 from the nitrogen fertilised plots and 21 from the plots fertilised with NPK. Two-way ANOVA on the run-off nitrate concentration data revealed that fertiliser treatment had no significant impact, with variation in date of measurement alone being able to explain 52.01% of the variation in nitrate concentration. None of the covariates could explain any of the further variation in nitrate concentration.

4.3.5.4 Chloride concentration

A total of 182 samples were analysed for chloride concentration over the trial period, consisting of 49 from the control plots, 51 from the plots treated with basic slag, and NPK, 34 from those fertilised with basic slag, 25 from those fertilised with nitrogen and 23 from the plots fertilised with NPK. Two-way ANOVA on the run-off chloride concentration data revealed that fertiliser treatment had no significant impact, with variation in date of measurement alone being able to explain 37.78% of the variation in chloride concentration. Inclusion of the covariate run-off water pH found that this could further explain some of the variation in chloride concentration, however none of the other covariates were found to have a significant impact, and the insignificant affect of fertiliser treatment remained. Together, date of measurement and run-off water pH could explain 55.3% of the variation in chloride concentration.

4.3.5.5 Phosphate concentration

Statistical analysis revealed that none of the variation in phosphate concentration could be explained by variation in fertiliser treatment; however variation in the date of measurement could explain 21.31% of the variation in phosphate concentration.

4.3.6 Initial and final soil pH

Soil pH measurements made on soil samples taken from the plots before fertiliser application ranged from a mean of 5.52 to a mean of 5.78. The lowest mean value of 5.52 was found below one of the plots to be fertilised with basic slag, and the highest mean value of 5.78 from one of the plots to be fertilised with NPK. Statistical analysis via one-way ANOVA revealed no statistically significant difference between the mean pH of any plots, therefore initial soil pH was not considered as a covariate in any further analysis. This meant that as with %SOC the post trial control and fertilised plots could be compared to reveal the impacts of grassland fertilisation.

Post trial analysis of soil pH revealed a statistically significant difference between plots, ranging from a mean of 4.92 to 5.41, with the lowest pH found under a plot which had been fertilised with nitrogen, and the highest under a plot fertilised with basic slag. One-way ANOVA was then undertaken to compare fertiliser treatments rather than individual plots, revealing a statistically significant difference in soil pH. Post-hoc analysis indicated that the significant difference occurred between plots fertilised with basic slag and those fertilised with nitrogen, with the basic slag treated plots have a significantly greater soil pH. This variation in soil pH one year after fertiliser application is illustrated in figure 4.17.

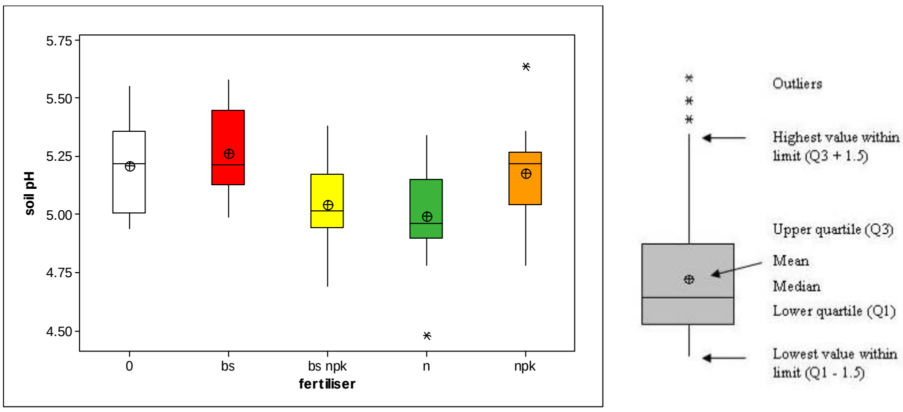

Figure 4.17 The variation in soil pH one year after fertilisation. Where 0: control (no fertiliser); bs: basic slag; bs npk: basic slag and NPK; n: nitrogen; npk: NPK

All pre and post-trial soil pH data is located in Appendix 6 under the heading 'Soil pH'.

4.4 Discussion

The lack of a significant difference in %SOC between plots treated with different types of fertiliser one year after fertilisation could initially be concluded to show that fertiliser treatment of grasslands has no affect on %SOC. Comparison of pre and post-trial %SOC under each respective treatment does though indicate that %SOC under some treatments has declined to a greater extent than under other treatments, with no significant decline under the nitrogen fertilised plots. This suggests that the application of nitrogen fertiliser may be responsible for maintaining %SOC, and that it is depleted with the use of other fertilisers, and when left un-treated. The issues discussed in Section 4.2.7 however, suggest that changes in %SOC are difficult to detect, highlighting a need for much longer trials, where fertiliser is applied to plots on an annual basis, and %SOC re-evaluated after a greater number of years. Although the results do not suggest that SOC accumulation has been promoted under plots fertilised with phosphorus based fertilisers, it is still possible that root growth has increased, however this was not measured in this trial and more research is needed before this can be concluded. If measured, information regarding root growth could then be used to predict how %SOC may respond, and allow some results to be gained over a shorter time period than that needed to detect a change in %SOC. It is important to note also, that even though the results of this trial do not correlate with those of Chapter 3, the nature of the questionnaires in Chapter 3 meant that they were unable to control for interactions, including that between fertiliser and grazing, making it possible that grazing intensity under phosphate fertilised plots, rather than the fertiliser itself was responsible for the apparent higher %SOC. The observed effects of fertiliser treatment in this chapter may not be contradictory with the results of Chapter 3, but instead it may be that interacting factors including fertiliser application and grazing intensity need to be studied in more detail. It is also possible that interactions between fertiliser and grazing may have taken place in this current trial, as grazing intensity was not monitored on plots varying in fertiliser treatment. The consequences of fertiliser application on grazing intensity have been noted in other studies (e.g. Schipper et al., 2009), and require much further research.

Although the results of the one year trial have revealed no significant difference in the NER or PP of grasslands under different fertiliser treatments or unfertilised grassland, a significantly greater NEE under nitrogen fertilised plots than un-fertilised plots was observed. Despite suggesting that the application of nitrogen fertiliser will increase carbon sequestration, in contradiction to the results of Chapter 3, it is again essential to realise that the impacts of such fertilisation on these gaseous carbon fluxes may require a longer length of time to emerge. This is emphasised in research by Davies (2006), where a phosphate fertiliser (Gafsa) was found to underperform other fertilisers in year one of the study, but to then produce the most productive results in years two and three. As with the impacts on %SOC, the impacts of fertiliser and grazing on NEE need also to be considered, due to grazing intensity being an uncontrolled variable in this trial. This suggests that no firm conclusions can be drawn from this chapter after only one year, and that NEE, NER, PP and %SOC of grasslands should be re-examined following several more years of current fertiliser treatment.

A lack of a significant difference in total above-ground biomass accumulation over the one year period following fertilisation does not agree with the findings of Hati et al (2008), who found increased crop biomass following fertilisation; however their research was specific to arable land. Again, as with %SOC, NEE, NER and PP the findings from this chapter can not be considered conclusive until a longer running trial is undertaken, allowing the long term effects on all aspects of the carbon cycle to be revealed. Firmer conclusions regarding the impacts of fertilisers on above-ground biomass could also be established in a trial where grazing is controlled, to avoid the issues of fertiliser/grazing interactions discussed above in relation to %SOC and NEE.

In relation to the impacts of fertiliser application on DOC concentration, there appears to be no significant impact on either the soil or run-off water. The main control on DOC concentration in this study was found to be time of year, with a significantly greater flux in the spring and summer than winter months. The results of this trial do not agree with those of McTiernan et al (2001) from undrained soils, where significantly greater fluxes were observed under nitrogen fertilised plots, however they support their findings of no change with fertilisation on drained soils. As McTiernan et al (2001) attribute the increased fluxes from un-drained nitrogen fertilised plots to increased dry matter production, increased root and leaf death, and increased OM production, the lack of any significant difference in dry matter production in this study (Section 4.3.2) could explain the lack of a significant difference in DOC. The same cautions as those taken when analysing the impacts on %SOC, NEE, NER and PP must however be considered, as the effects of fertiliser treatment on soil and run-off water DOC over one year will not necessarily become apparent over such a short time period.

In relation to the environmental cost aspects of fertiliser application in addition to those on the carbon cycle, a significantly higher soil pH found under plots treated with basic slag than with nitrogen corresponds with the findings of Hati et al (2008) and Sharma and Subehia (2003). This suggests that soil pH will respond after only one dressing of fertiliser, and the results from this study confirm that basic slag will act as a liming agent and increase soil pH, whilst nitrogen application will result in slight soil acidification. The significant liming ability of basic slag and acidifying affect of nitrogen fertiliser has not though translated into changes in grassland productivity as observed in other studies (Hati et al., 2008; Sharma and Subehia, 2003). The significant increase in soil pH under basic slag does however suggest that grassland biomass yields could increase in the future, and the decrease in pH under nitrogen treated plots suggests that grassland biomass yields could decrease. Although this provides some support for the hypothesis of increased %SOC under phosphate fertilised grassland, it again emphasises the necessity for longer running fertiliser trials.

In addition to environmental costs on soils and the carbon cycle it was a major aim of this chapter to assess the impacts of grassland fertilisation on run-off and soil water chemistry. The results of this study suggest that other than a reduction in soil water pH under NPK and nitrogen fertilised plots, there would also be greater nitrate and chloride losses from grasslands treated with a combination of NPK and basic slag. The decline in soil water pH under plots treated with nitrogen could be the result of a release of hydrogen atoms during nitrification, however the reasons for increased nitrate and chloride concentrations under the basic slag/NPK fertilised plots are un-clear, and require further investigation. It is important to note also that although nitrate concentrations increased under NPK/basic slag treated plots this was an increase relative to plots treated solely with basic slag, with no significant increase over those left unfertilised. A lack of any significant change in any of the components of run-off water chemistry and no significant change in soil water phosphate concentration and nitrate concentration relative to the control suggests that the aim to cause minimum environmental impact to watercourses could be met with the application of any discussed fertiliser. The issues of assessing such change after only one application of fertiliser must however again be brought to attention, and further work should be undertaken to assess the impacts on water quality over longer time periods, and following several more annual fertiliser dressings. With regard to the impacts of fertiliser application on the pH of soil water a significantly lower pH beneath plots fertilised with nitrogen and NPK than below the un-treated plots and those fertilised with basic slag corresponds with the results found in relation to soil pH. This indicates that the use of nitrogen and NPK fertiliser not only lowers the pH of grassland soils, but also that of soil water. Although this effect must be taken into consideration when assessing the use of fertilisers on managed grasslands, due to groundwater acidification and the environmental concerns of the NT (Section 1.1.2), it must be noted that the decrease in pH was very small (Figure 4.12), declining from 6.93 to 6.78.

Although the use of basic slag on grasslands did not increase %SOC in this study, it is important to also consider the carbon emission costs of producing fertilisers, and the fact that these could reduce or offset some of the carbon benefits they bring (Purukayastha et al., 2008). The maintenance of %SOC under nitrogen fertilised plots must therefore be considered alongside the carbon emissions associated with production of such a mineral fertiliser. One of the main aims of the NT to reduce the use of artificial fertilisers (Section 1.1.2) must be considered, and although basic slag has not been shown to sequester carbon into grasslands additional to that sequestered under nitrogen fertilised plots, the organic nature of basic slag must be recognised. Measurement of the impacts on the complete carbon budget are beyond the scope of this chapter, however it is a necessary area of further research before a firm conclusion regarding fertiliser application to grasslands can be made.

4.5 Conclusion

The results of a plot trial in north east England where four varieties of fertiliser were applied to a managed grassland in the summer of 2009 reveal no significant difference in NER, PP or run-off water chemistry ($p < 0.05$). A decline in %SOC following fertilisation under all treatments other than the nitrogen fertilised plots suggests that %SOC could be maintained with nitrogen application. A significantly greater NEE under nitrogen fertilised plots relative to the control plots also suggests that nitrogen fertilised grasslands have the potential to sequester greater amounts of atmospheric carbon. Although the results of this trial do not support the findings of Chapter 3, which hypothesised accumulation of SOC with basic slag application, it must be realised that the impacts of fertilisers on all aspects of the carbon balance may take longer than 1 year to occur, highlighting a requirement for longer term trials following annual fertilisation. In addition, the application of different fertilisers in this trial was found to have no effect on soil water DOC, nitrate concentration or phosphate concentration relative to un-fertilised land, however a difference in soil water pH and chloride concentration was observed. A significantly lower soil water pH was found beneath plots fertilised with nitrogen and NPK than below un-treated and basic slag fertilised plots, and a significantly greater chloride concentration in soil water under plots fertilised with a combination of basic slag and NPK. Even though the observation of a change in soil water pH is in agreement with the impacts on soil pH, which showed an increase beneath plots fertilised with basic slag relative to unfertilised plots, and a decrease in the soil pH of plots fertilised with nitrogen, it must be recognised that the extent of the change in pH is small. These results suggest that fertiliser variations will not have any major impacts on the soil and run-off water draining into surface waters and streams, and that the aims of the NT to have minimum environmental impacts will not be altered to a large extent by varying the type of fertiliser applied.

All results from this trial must however be taken with caution, and further work is necessary to establish whether these immediate effects are representative of fertiliser application on an on-going annual basis. The findings of increased soil and soil water pH below plots fertilised with basic slag suggest that this fertiliser has a liming affect, and that there is potential for increased biomass stocks and increased %SOC. This cannot though be considered a firm conclusion until similar trials are undertaken over a much longer timescale. Complete assessment of fertiliser production carbon emissions are also required before guidelines regarding the most appropriate grassland fertilisers to apply are drawn up.

Chapter 5

Testing the transferability of results to other NT landholdings: The Wimpole validation site

5.1 Introduction and Rationale

A major aim of the soil sampling at Wallington (Chapter 2) was to assess the potential to estimate a region's SOC stocks without such extensive soil sampling as that undertaken on the Wallington Estate. The Wallington Estate was initially chosen as a location to measure SOC stocks due to its extensive range in altitude, soil type and land-use, meaning therefore that the majority of soil-types, land-uses and altitude ranges covered by other NT estates across the UK would be represented by the soil samples taken at Wallington.

In an ideal situation areas of high SOC storage could be identified and preserved anywhere in the UK by simple reference to soil maps or land-use maps (or a combination of both) and their respective assigned %SOC values, without the need to spend a large amount of time undertaking fieldwork across the country. The results of Chapter 2 however, show that there are limitations with these methods, and that relying on published soil maps and memoirs detailing typical soil group or soil series %SOC, will not provide the most accurate of SOC stock estimates. Chapter 2 indicates that relying on land-use maps and published %SOCs for typical land-uses will neither give the most accurate of estimates, nor will knowing both the land-use and soil series of the areas in question. The conclusions drawn in Chapter 2 indicate that more specific information relating to altitude and soil pH will greatly improve any %SOC estimates, as will knowing the specific farm from which the soil samples are taken. The identification of farm as a control on %SOC suggests that a large amount of variation within grazed pasture may be the result of differences in land-management practices between farms.

The results of Chapter 2 suggest that fieldwork and sample collection are necessary to produce the most accurate of %SOC estimates, and that this is an improvement on using NSRI and CSS mean %SOC values assigned to particular land-uses or soil groups/series. There is a possibility, however, that the mean %SOC values found at Wallington for major soil group, soil series, land-use and their combinations may produce accurate estimates of %SOC at other locations across the UK, and that very accurate estimates could be produced if the statistical models found to explain the greatest amount of variability at Wallington are used to predict the %SOC at other locations. If this is true it will mean that extensive fieldwork need not be undertaken.

The aim of this chapter was to collect and analyse soil samples from another NT estate, and produce %SOC maps and identify %SOC values for point locations covering a variety of land-uses. %SOC values from the Wallington Estate could then be used to produce predicted %SOC maps for the new estate, and %SOC values for all soil sample locations. The predicted %SOC maps could then be compared to the %SOC maps produced by %SOC analysis of soil samples, and the predicted %SOC values can be compared to those found in the field to identify the accuracy and precision of using the Wallington dataset to predict %SOC values at other locations.

The objectives of this chapter were to:

1.) Establish whether %SOC predictions for other areas of NT land in the UK were also more accurate when altitude was used as a predictor in addition to soil type and land-use.
2.) Establish the degree to which land-management was having an effect on other NT estates by assessing the variation in %SOC values beneath individual land-uses, and identify whether %SOC predictions improve with farm tenancy included as a factor in the model, or whether any new factors can be found to explain variability that were unidentified at Wallington.
3.) Establish whether %SOC values found at Wallington for specific soil types and land-uses would provide a more accurate prediction than those from secondary sources such as NSRI and CSS, where data may have been collected several decades ago.
4.) Establish whether the statistical models found to explain the greatest amount of variation in %SOC at Wallington can be used to produce the most accurate estimate of %SOC at other locations when the option of soil collection and analysis is not available.

5.) Establish the most important sources of information needed to calculate the %SOC values for other areas of NT land based on the results of both this work and the results of Chapter 2.

5.2 Materials and methods

In order to determine how accurate the results gained from the Wallington soil campaign were, in terms of predicting the SOC stocks for other estates across the country, it was decided to carry out a similar sampling campaign on an estate in a very different area of the UK, with a different climate and with different soil series (although preferably similar soil groups). It was also essential that the estate had the same land-use classes as at Wallington. The Wimpole Estate in Cambridgeshire was therefore chosen.

5.2.1 The study site: Wimpole

The Wimpole estate is located in eastern England, in the county of Cambridgeshire, where climate and soils are different to those of NE England and the Wallington pilot study site. The Estate covers a total area of approximately 1000 ha, 660 ha of which is used for arable farming, and 240 ha are under permanent grassland and have received no treatment in the form of fertilisers over the last 10 years. 100 ha of the Estate are covered by woodland which was under arable land-use prior to woodland establishment. The location of the Wimpole Estate is shown in Figure 5.1.

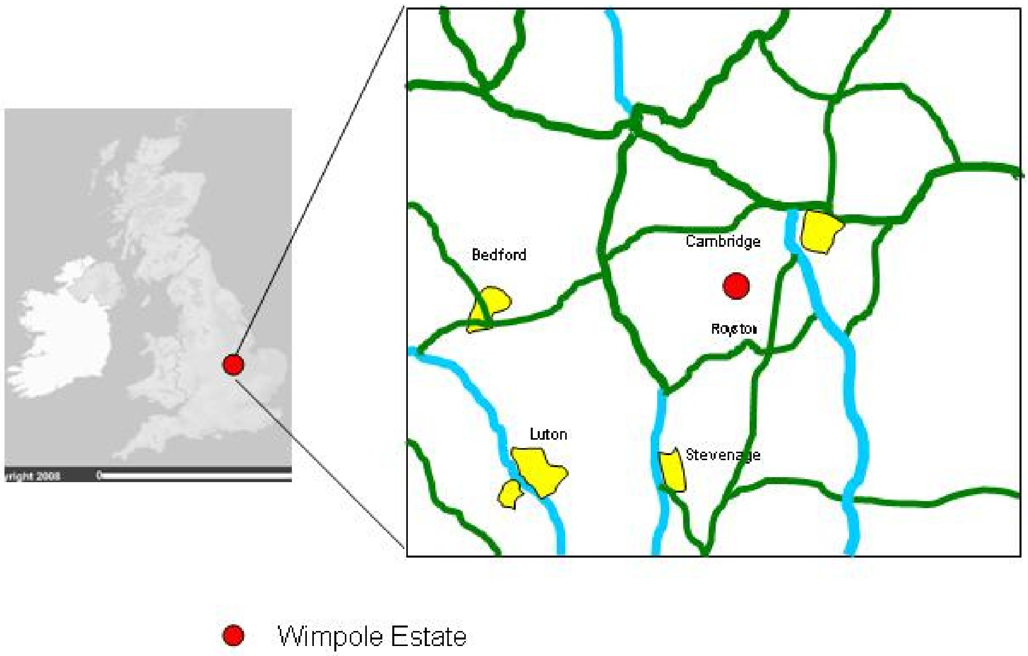

Figure 5.1: The Wimpole Estate, Cambridgeshire

The range and location of major soil groups, land-uses and farms, as well as altitude variation across the Wimpole Estate can be seen in Figure 5.2.

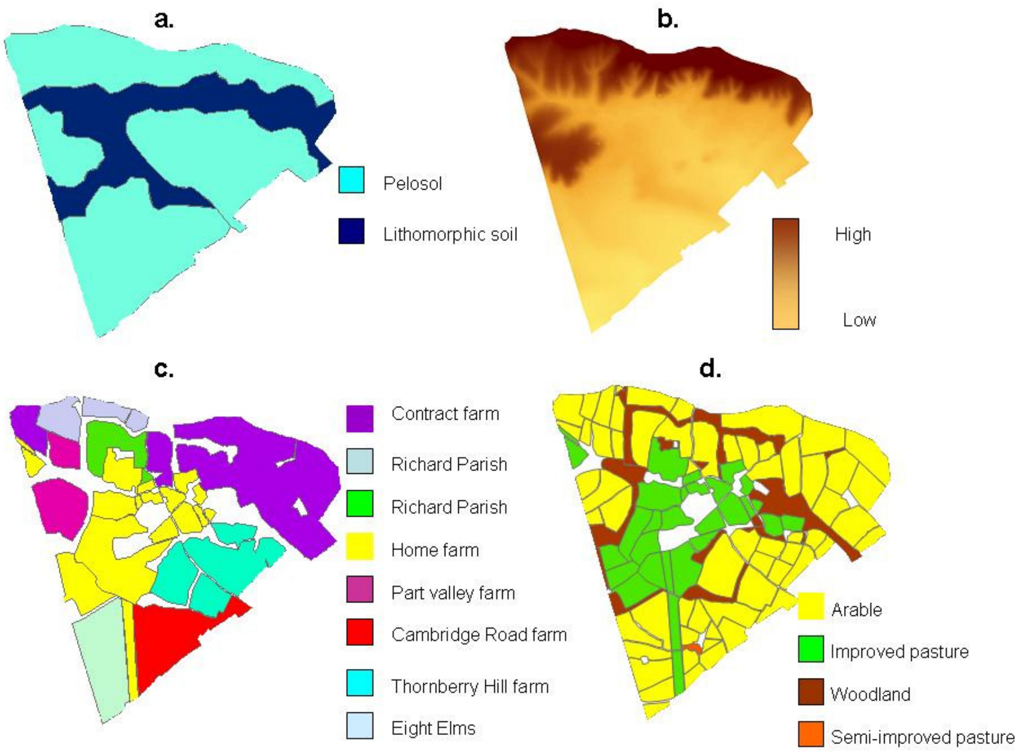

Figure 5.2: a. Wimpole major soil group distribution; b. Wimpole altitude range; c. Wimpole farm locations; d. Wimpole land-use distribution

5.2.2 Estimate of %SOC values using soil samples

A total of 378 soil samples were collected from the Wimpole Estate in May 2009 following the methodology described in Section 2.2.2. The land-uses at Wimpole were classified into the following categories (land-use class 1): *Arable*, *Improved pasture*, *Semi-improved pasture* and *Woodland*. As a result of the findings from Chapter 2 which suggest land-management practices as a reason for variation in %SOC within a land-use, a second classification scheme (land-use class 2) was created. In land-use class 2 the land-use *Improved pasture* was further classified into *hay meadow* and *Improved pasture,* based on information regarding grassland management provided by the Estate manager. This was done to assess the possible impacts of grassland management on %SOC, and to determine whether more accurate SOC stock estimates could be produced knowing such additional information. Unlike the Wallington Estate there was however no NT Biological Survey available for Wimpole, therefore land-use maps provided by the farm manager were used as an initial guide to land-use classification, supplemented with subjective observation in the field. Although the results from Chapter 2 suggest pasture land-management techniques as a potential factor causing variations in %SOC, the role of fertiliser management could not be investigated due to all pasture at Wimpole having received no fertiliser in the past 10 years. This meant that pastures could not be classified further into those treated with different fertiliser.

The Wimpole Estate boundary was entered into ArcGIS and the NSRI map of Great Britain (Avery, 1980) was used to create feature classes for major soil group, with the two major soil groups being identified as Lithomorphic soils and Pelosols. Feature classes for land-use and farm tenancy were created using ordnance survey maps as a guide. The mean %SOC values from the 378 soil samples were then calculated for each major soil group, land-use category and farm tenancy respectively (as described in Section 5.2.2.1). These values were applied to the area of the feature class which they represented.

5.2.2.1 Analysis of %SOC

Chemical analysis of %SOC was undertaken using the methodology described in Section 2.2.2.1. A further quality control check to assess the accuracy of both the loss-on-ignition and Walkley-Black methods was undertaken on a sample of the Wallington soils before analysis of the Wimpole soils. This was done to assess the accuracy of both methods, and to provide confidence in the %SOC values attained from such methods. This quality control check involved CHN analysis of a sample of Wallington soils. CHN analysis involved measurement of the content of each of the elements carbon (C), hydrogen (H) and nitrogen (N) using a COSTECH ECS 4010 Elemental Combustion system with a pneumatic autosampler at the University of Durham. This method involved two reactors, one consisting of chromium (III) oxide/silvered cobaltous-cobaltic oxide catalysts at 950 °C and the other of reduced high purity copper wires at 650 °C. The gases were separated using a packed (Porous polymer, HayeSep Q) 3m GC column, with Helium at a flow rate of 95 ml /min used as the carrier gas. The column was replaced after approximately every 120 samples, and the signal of each sample was measured using a thermal conductivity detector. Repeats of every sample were run for the purposes of quality control, as were laboratory standards in every sample run. The laboratory standards used were BBOT (a calibration standard, COSHTECH Analytical Ltd) and a high organic standard (b2150, Elemental Microanalysis Ltd). The calibration between values produced using the Walkley-Black method and those using CHN analysis is displayed in Figure 5.3. The r^2 value of 0.98 shows a very strong correlation between methods, and indicates that the Walkley-Black method of TOC analysis can be used with confidence when analysing %SOC. A lower value from the Walkley-Black method is to be expected as this is a measure of organic carbon compared to CHN analysis which measures total carbon in the sample. Any inorganic carbon in the sample would be included in the measure of total carbon and the inorganic carbon content of soils is often substantial in soils containing large amounts of $CaCO_3$ (Tan, 1996). Although the inorganic carbon content of soils was not measured in this study, nor is it detailed in the NSRI representative soil profile descriptions of soils, it is likely to exist in soils of a calcareous nature (Tan, 1996). Such soils do exist on the Wallington Estate (Payton and Palmer, 1990), however no inorganic carbon is likely to exist in the acid non-calcareous soils on the Estate (Tan, 1996), indicating that the Walkley-Black method used in this study may be slightly under-estimating %SOC.

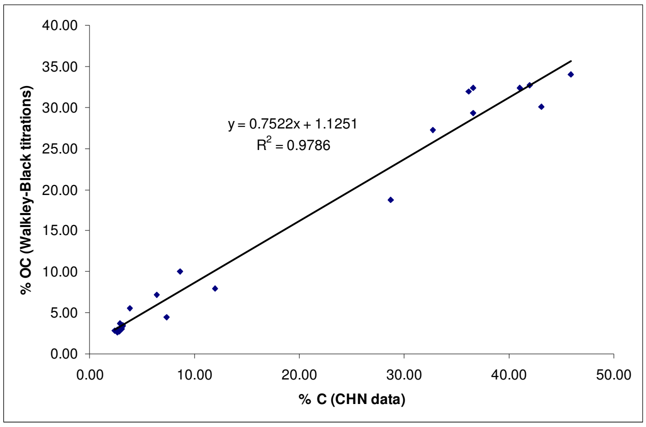

Figure 5.3: The relationship between %OC analysed by the Walkley-Black method and %C by CHN analysis

5.2.2.2 Soil pH

Soil pH of all soil samples was measured as in Section 2.2.2.2.

5.2.2.3 Land-use History

As in the Wallington study a detailed land-use history was required to assess which soil carbon pools are in equilibrium and which are adjusting to previous land-use change (Stevens and Van Wesemael, 2008). Although land-use history was found to be an insignificant control on %SOC on the Wallington Estate the majority of land has undergone very little land-use change in the past 30 years. This lack of land-use change may be obscuring any effects on %SOC which may be the result of land-use history, and the fact that the effects of land-use history have been investigated or even found to cause variations in %SOC in other studies (Schulp and Veldkamp, 2008; Sonneveld et al., 2002; Conant et al., 2003; Venteris et al., 2004) suggests that it is a vital factor to include in any analysis of %SOC. A limited amount of time however meant that detailed land-use history surveys and interviews with the tenant farmers regarding their land-use over the past 30 years could not be undertaken. Land-use history information provided by a forester and Estate manager, both of whom had detailed background knowledge of the Estate was therefore utilised. As with the Wallington study, limitations regarding the reliance on such secondary data must be taken into consideration.

The following land-use history classes were created with the current land-use and any previous land-uses recorded as follows: A was A (arable land that has been in constant arable land-use over the past 30 years), A was IP (arable land that was improved pasture before it converted to its current land-use), H was H (fields currently cut for hay that have been in this land-use for the past 30 years), H was A (hay fields that were in arable land-use before they were converted to their current land-use), IP was IP (Improved pasture that has always been improved pasture), IP was A (improved pasture that was arable before being converted to its current land-use), SI was SI (semi-improved pasture that has always been semi-improved pasture) and W was W (woodland was woodland).

5.2.2.4 Water content

As with the Wallington soil samples, soil water content was not measured in this study. This was due to the time period over which the soils were sampled and subsequently analysed, and the possibility that water content would be more indicative of weather conditions on the day of sample collection. Soil water content was therefore not included in any analysis of %SOC.

5.2.3 Estimate of %SOC using published soil survey data and maps

5.2.3.1 NSRI data

A 1:25,000 scale soil map of the Wimpole Estate does not exist and so could not be used to identify individual soil series as in the Wallington study. Although a 1:25 000 scale map has been produced for the Royston area (sheet TL34, Seale, 1986), the large majority of the Estate is not included in this map. Identification of soils on the Wimpole Estate therefore relied on use of the Great Britain 1:250,000 scale map (Avery, 1980), which enabled identification of the two dominant soil series present on the Estate: Wantage and Evesham/Hanslope. Reference to a report undertaken by the NSRI (Burton, 2004) indicates that greater than seven individual soil series could be present on the Estate; however a lack of a published map providing the aerial extent and location of each of these has meant that the 1:250,000 scale map has had to be used. The soil map was scanned onto a computer and digitised as in Section 2.2.2. All soil samples were then assigned a %SOC value for the particular soil series from which they were taken as according to the NSRI map. The %SOC values applied to each soil series were taken from NSRI memoirs indicating typical %SOC for individual soil series (Seale, 1986).

5.2.3.2 CSS data

The maps produced in Section 5.2.1 were used to identify the location of each sample point in relation to its land-use and major soil group. Mean land-use and major soil group %SOC values from the CSS database were then applied to each sample location as in Section 2.2.3.2 .

5.2.4 Estimate of %SOC using Wallington data

The location of all soil sample points relative to land-use and soil type was identified by reference to the maps produced in Section 5.2.1. Due to the large range in soil groups, soil series, land-uses and altitudes sampled at Wallington (chapter 2) it was hoped that these variables on any other NT estate in the UK would fall within the ranges/categories sampled at Wallington. This would then allow the mean %SOC values for land-use, major soil group and soil series, and soil series/land-use combinations to be applied to other estates to accurately predict %SOC values at point locations, and to identify areas of high carbon store worth protecting, and areas of low carbon store where there is potential to make land-use change to increase stocks. The use of these Wallington %SOC values to estimate other NT SOC stocks can not however proceed until it is known how well they can predict %SOC in locations other than Wallington. As seen in Chapter 2 the most accurate %SOC values for the Wallington Estate were gained when mean values for a land-use/soil series/farm tenancy combination were applied to a soil sample taken from an area of land under that combination of variables and adjusted for altitude and soil pH. As discussed in Chapter 2, the farm tenancy element suggests an influence of land-management on SOC stocks, which may in the future enable further stratification of land-uses into those which have been fertilised or grazed in different ways, despite still being classified as the same land-use. As these land-management factors have yet to be established however, it was not possible to use this variable in any predictions of SOC stocks on other NT estates, and therefore the variable farm tenancy (indicative of land-management) could not be used as a predictor of SOC stocks. In a similar manner, although grassland cut for hay was being compared to continuously grazed pasture to identify if it is a control on %SOC at Wimpole, the results from the Wallington Estate could not be used to apply different %SOC values to these different types of grassland, as grassland was not stratified into these land-use classifications during the Wallington campaign.

To apply the regression equation from the GLM explaining the greatest amount of variation in the data on the Wallington Estate to the Wimpole Estate would require knowing the soil series, the land-use, the altitude, the soil pH and the farm of each location at Wimpole. Not only would this information need to be known, but the soil series, land-use and farm would have to be ones from which samples had been taken at Wallington. If this data was available and the Wallington model was valid in other locations (e.g. Wimpole), then the use of the regression equation should produce predicted %SOC values similar to the observed %SOC values. To establish the accuracy of the Wallington data when applied to the Wimpole Estate, regression equations were constructed using the coefficients for the factors and covariates from the Wallington model. The predicted %SOC values calculated from the regression equations were then compared to the observed %SOC values from soil sample analysis and statistically compared using a paired t-test (see Section 5.2.5). Visual analysis of the accuracy of the Wallington data in predicting %SOC at Wimpole was also made possible by constructing %SOC maps by inverse distance weighting interpolation of all point values of %SOC in ArcGIS.

A lack of information and knowledge as to the reasons behind the influence of farm at Wallington means that the Wallington model with the greatest predictive accuracy could not be used. The next best models therefore had to be used, however the models found to explain the greatest amount of variation in %SOC at Wallington nearly all included soil series as a factor (Chapter 2). Since none of the soil series present at Wallington are present at Wimpole, none of these models could be used to predict the %SOC at Wimpole locations. Regression models were therefore created from the Wallington data, using the factors and covariates that were in existence at Wimpole. All land-uses present at Wimpole had been sampled at Wallington, however only one of the two major soil groups present at Wimpole had been sampled at Wallington. The altitude range at Wimpole spanned 23- 106 m and is therefore not covered by the range of altitudes sampled at Wallington (see Chapter 2). The major soil group *Lithomorphic* soils had been sampled on the Wallington Estate and so the coefficient for this soil group from the Wallington data (Chapter 2) was used to predict %SOC values for soil under this major soil group at Wimpole. The major soil group *Pelosol* had not been sampled at Wallington due to no such soil type existing on the Wallington Estate. Coefficients from the models used to predict %SOC at Wallington using the major soil group *Brown soils* were therefore used at Wimpole for all sample locations under *Pelosols,* as reference to soil memoir and classifications (Avery, 1980) suggest this as the most comparative major soil group.

Based on the Wallington results, it was thought that the most accurate predictions of %SOC would be achieved if mean %SOC values for the major soil group/land-use combinations were applied to the specific point in question- if no other fieldwork data had been carried out or soil property information was available. Although inclusion of altitude in the Wallington model led to greater predictive accuracy it was uncertain whether using altitude coefficients from the Wallington Estate would improve the predictions at Wimpole due to the altitude range being un-sampled at Wallington.

5.2.5 Statistical Analysis

The sampling design conducted within this study could be considered as a three factor experiment with multiple covariates. The three factors were: soil series/major soil group; land-use/land-use history and farm tenancy. All three factors were entered into a GLM as categorical variables using Minitab statistical software. The covariates considered were: altitude, soil pH, and aspect. This meant that the data could be analysed by ANCOVA. Results were considered statistically significant if $p < 0.05$ (95% confidence interval). The results of ANCOVA were post-hoc tested using the Tukey test, and proportion of the original variance explained by factor and covariate was calculated using the method of Howell (1996). Descriptive statistics were used to compare the variability within the different levels of soil or land-use classification. The r^2 values generated by ANCOVA represent the between *group sum of squares* divided by the *total sum of squares,* with a large r^2 value thus indicating that a large fraction of the variation in the independent variable can be explained by the categorical variable/treatment. The r^2 value represents the proportion of the total variation explained by the difference in the means.

The predictive value of using NSRI and CSS datasets for predicting %SOC based on major soil group/soil series or land-use was analysed using paired t-tests. The same paired t-test method was used to analyse the predictive value of using mean values for land-use or major soil group from the Wallington study to predict %SOC based on either land-use, major soil group, land-use/major soil group combinations, or land-use/major soil group corrected for altitude. Paired t-tests allowed the difference between every measurement and the predicted measurement for that location to be viewed and to then reveal how the mean value of the actual measurements differs from the mean value of the predicted measurements. The t-value indicates the mean difference between the predicted and actual values and whether this is significantly different from zero. A t-value of zero would indicate no significant difference between the predicted and actual %SOC values, and suggest that the database used could be used confidently to predict %SOC values for locations at Wimpole. The confidence interval for the mean was also assessed along with each t-value, and if this was found not to include zero then it could be confidently stated that the predictions were significantly different to the measured %SOC.

5.3 Results and discussion

All %SOC data from the Wimpole Estate can be found in Appendix 2 under the heading 'Wimpole soil samples'.

5.3.1 Controls on %SOC

5.3.1.1 Stratification into soil series/major soil group

As with the Wallington study the ability to predict %SOC at point locations knowing only the major soil group or soil series of the land area in question was assessed initially. As explained, the soil map of 1: 125,000 scale meant that only two soil series were identified, each of a different major soil group, therefore the predictability in this study of using major soil group is the same as that of using soil series. The results of this analysis revealed that only 1.22% of the variation in %SOC could be explained by the variation in major soil group/soil series (when soil series is mapped at such a scale). The coefficients of variation of 42.34 and 42.56 for Lithomorphic and Pelosols respectively show an extensive amount of variation within both soil groups, indicating that many other factors in addition to major soil group are responsible for %SOC variation. Although the difference in %SOC between major soil groups was statistically significant, the fact that only 1.22% of the variation in %SOC can be predicted knowing only major soil group indicates that this is not sufficient information on which to predict SOC stocks, supporting the results of the Wallington study. The results of this sampling campaign indicate a mean %SOC for Lithomorphic soils of 3.25% and a mean %SOC for Pelosols of 2.91% (Figure 5.4). As with the Wallington study a large range in land-use and altitude beneath each respective soil series (Figure 5.2) is likely to be responsible for much of this variation.

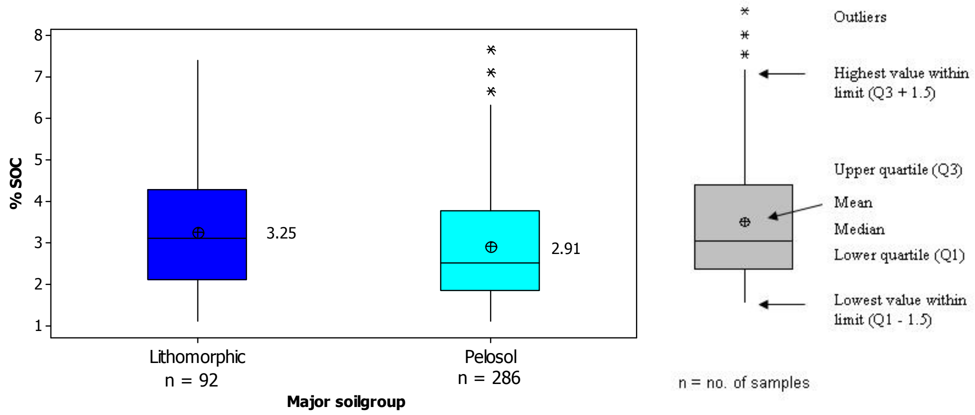

Figure 5.4: The variation in %SOC beneath major soil groups

5.3.1.2 Stratification into land-use (Land-use class 1)

Smaller CVs ranging from 19.18 to 27.63 indicate less variability in %SOC beneath individual land-use classes when classified in this way than beneath major soil groups/soil series. An r^2 value of 59.23% and a significant difference in %SOC between land-uses indicates that SOC stock estimates can be made much more confidently knowing a mean %SOC value for land-use rather than major soil group. Analysis of %SOC reveals a statistically lower %SOC under arable than all other land-uses, a statistically lower %SOC under improved pasture than under woodland, however no statistical difference in %SOC under semi-improved pasture and any other land-use (Figure 5.5). Reference to Wimpole Estate land-use maps (Figure 5.2) reveals less variation in other variables beneath individual land-uses than beneath the major soil groups/soil series, suggesting therefore that the greater predictability using land-use is to be expected if other factors such as altitude also have an influence on %SOC.

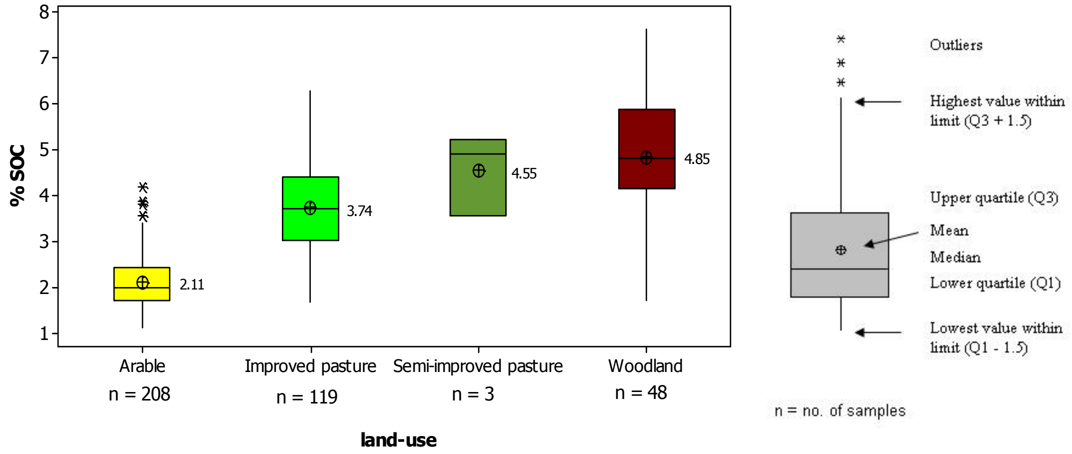

Figure 5.5: %SOC under land-uses at Wimpole

5.3.1.3 Stratification into land-use (Land-use class 2)

Larger CVs ranging from 19.18 to 32.91 indicate greater variability in %SOC beneath some individual land-use classes when land-use is classified in this way as opposed to land-use class 1. Despite the larger CV the r^2 value of 60.15% and the statistically significant difference in %SOC between land-uses indicates that land-use class 2 is a better predictor of %SOC than major soil group, and that classifying the land-use Improved pasture further into Improved pasture and Hay meadow will increase the predictive value of any %SOC estimates when compared to estimates using land-use classification 1. Although there is a large CV of 32.92 for Hay meadow, the CV of 24.67 for Improved pasture compared to the earlier CV of 27.63 and the significant difference between %SOC values below Hay meadow and Improved pasture suggests that grassland management may have some degree of control on %SOC, however as with all one way ANOVA a lack of control in relation to other variables must be taken into consideration. The mean %SOC values under land-use class 2 are shown in Figure 5.6.

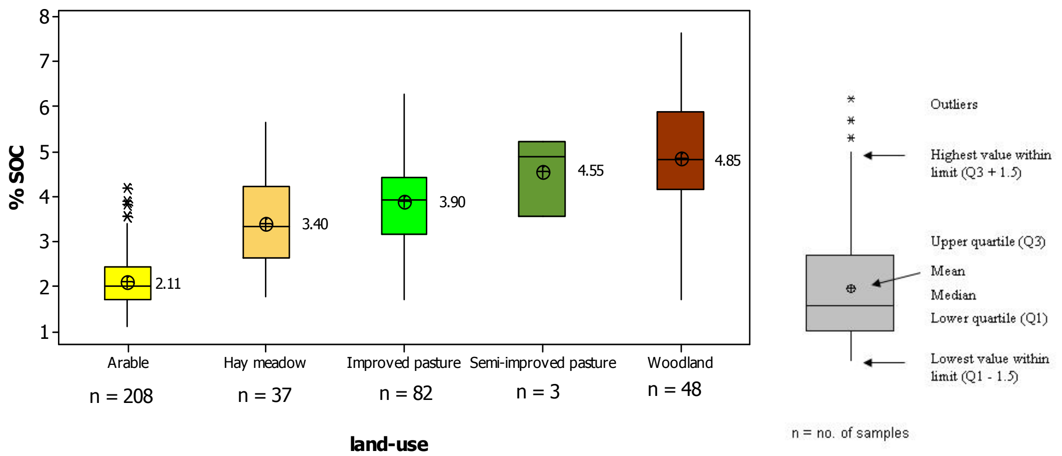

Figure 5.6: %SOC under land-use at Wimpole when classified as land-use class 2

5.3.1.4 Stratification into land-use (land-use history)

An r^2 value of 61.18% is an increase on all other r^2 values using either land-classification 1, 2 or major soil group in a regression model to explain the variation in %SOC. This indicates that the most accurate estimates of %SOC at point locations will be achieved if land-use is classified not only according to its current-use but also by its previous land-use. CVs ranging from 22.86 to 37.88 indicate that there is still a large amount of variation in some land-uses even when classified according to their current and past land-use, however this is again to be expected due to variations in other %SOC controlling factors beneath individual land-uses. The statistically lower %SOC under ‘A was A’ than any other land-use indicates that arable land that has never been laid down to grass has a significantly lower %SOC than arable land that has previously been grassland. A statistically insignificant difference between ‘A was I’ and ‘I was A’ suggests that both land-uses are still recovering from their previous land-uses and that their %SOC has not yet reached equilibrium. Although there was a statistically significant difference in samples taken from arable land-use depending on the previous land-use this was not true for any other land-use. There was no statistically significant difference in %SOC between ‘H was A’ and ‘H was H’ or between ‘I was I’ and ‘I was A’. These results are again however the result of one way ANOVA, therefore a lack of a significant difference could be the result of differences in other variables not accounted for if no information on these variables is provided. An insignificant difference between some land-uses could also be the result of the small sample size of some individual land-use classes (Figure 5.7).

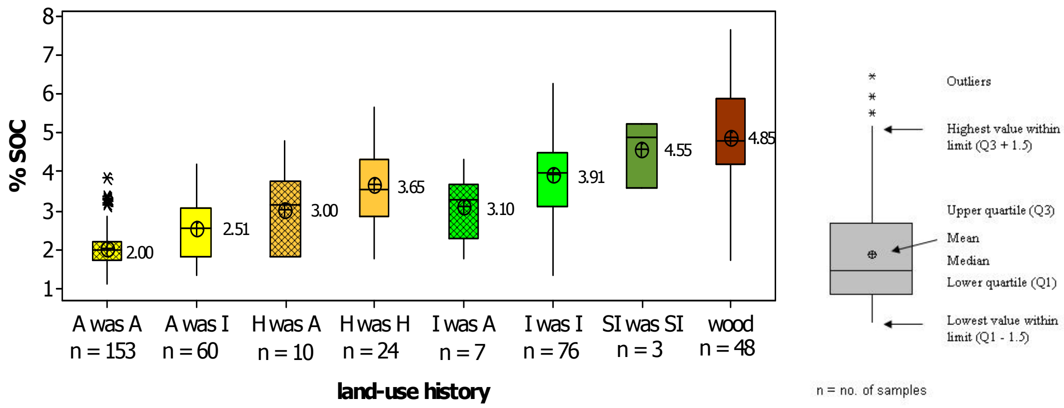

Figure 5.7: %SOC under land-use at Wimpole when classified according to present and past land-use.
A: arable; I: improved pasture; H:hay meadow; SI: semi-improved pasture; wood: woodland

5.3.1.5 Stratification into farm tenancy

An r^2 value of 52.54% suggests that knowing a mean value for %SOC beneath an individual farm location will produce a better estimate of SOC stocks than knowing a mean value for major soil group and applying this to the area of the major soil group. The r^2 value is however lower than that for land-use, indicating that applying the mean value for each land-use to the area of that land-use would produce a better SOC stock prediction than if you were to do this same task but using the mean value for farm tenancy. Although the CV’s beneath some farms are as low as 13.46%, other CV’s as high as 48.13 emphasise the variation in %SOC evident beneath other farms (Figure 5.8), and confirm that predicting SOC stocks using information relating only to farm tenancy could prove very inaccurate. As with the Wallington case study this is to be expected beneath those farms with a number of varying land-uses, a large aerial extent and range in soil groups and altitude (Figure 5.2).

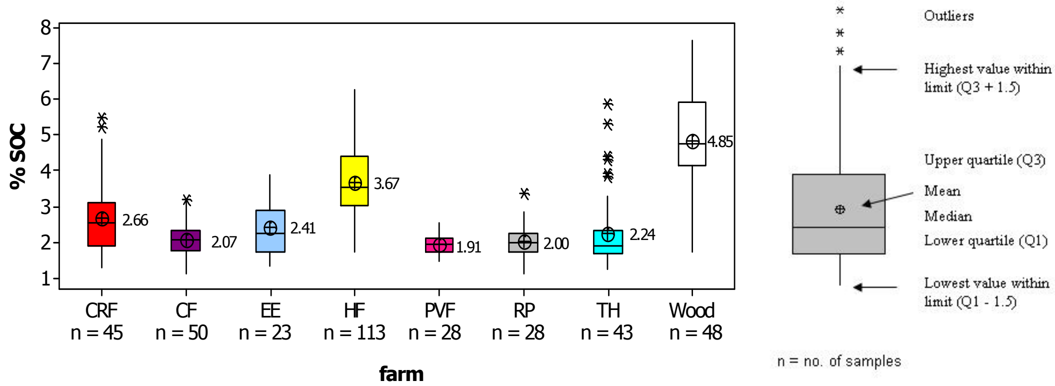

Figure 5.8: The variation in %SOC beneath farms at Wimpole
CRF: Cambridge Road farm; CF: Contract farm; EE: Eight Elms; HF: Home farm; PVF: Part Valley farm; RP: Richard Parish; TH: Thornberry Hill; Wood: woodland

5.3.1.6 Stratification into major soil group/land-use (land-use history)

An r^2 value of 61.40% when both the major soil group and land-use history are known indicates that estimates of SOC stocks will improve if a mean value for a land-use history on a particular soil group is applied to the area of that land-use/soil combination. When major soil group is included in the analysis along with land-use history the effect of major soil group does however become insignificant ($p>0.05$), indicating that under a constant land-use history although major soil group may have some degree of influence on %SOC the effect is not statistically significant.

5.3.1.7 Stratification into major soil group/land-use history/farm

An r^2 value of 61.52% when the effects of farm tenancy are included in the analysis alongside major soil group and land-use history, is a slight improvement on the 61.40% in variation that could be predicted if only major soil group and land-use history were known. Major soil group is again found to be an insignificant control on %SOC under a constant land-use and farm tenancy, and farm tenancy is also insignificant when the major soil group and land-use history are held constant. This suggests that under a constant land-use history there is no significant variation in %SOC regardless of which farm the soil sample was taken from. The insignificance of farm in this study suggests that in addition to the factors such as land-management practices and different fertiliser treatments, cropping management or grazing schemes which may be responsible for some of the variation between farms at Wallington, a more detailed land-use history or land-use history classification could be responsible for some of the variation ascribed to farm tenancy at Wallington.

5.3.1.8 Stratification into major soil group/land-use history/farm and altitude

An r^2 value of 63.73% when altitude is included in the analysis alongside major soil group, land-use history and farm indicates that altitude is also a major control on %SOC, and that some of the variation in %SOC under a constant major soil group, land-use and farm combination is the result of altitude. With altitude included in the analysis major soil group remains insignificant indicating that soils taken from beneath a constant land-use history and from the same farm and altitude will not vary in %SOC depending on the major soil group to which they belong. Farm tenancy also remains insignificant with altitude included in the analysis, indicating that soils taken from beneath a constant land-use history and from the same altitude and major soil group will not vary in % SOC regardless of the farm from which it was taken.

The insignificant role of farm tenancy and major soil group means that under a constant land-use history and altitude, these factors are not having a significant effect on %SOC and should not be included as factors in any model to predict future %SOC. Farm tenancy and major soil group were therefore removed from the model to leave a model with an r^2 value of 62.3%. This model indicates that the factors altitude and land-use history are those responsible for variations in %SOC on the Wimpole Estate. The variation in altitude and land-use can explain 62.3% of the variation in %SOC. The magnitude of effect of each of the factors is shown in Figure 5.9.

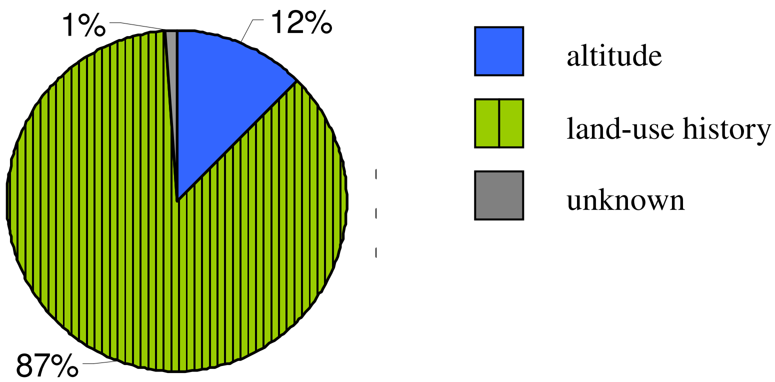

Figure 5.9: The magnitude of effect of land-use history and altitude on %SOC at Wimpole

5.3.1.9 Stratification into major soil group/land-use history/farm/altitude and pH

Inclusion of soil pH in the analysis did not result in any improvement in the ability to correctly predict %SOC at point locations on the Wimpole estate. Soil pH was found to have an insignificant effect on %SOC when major soil group, land-use, farm and altitude were constant. This result is in conflict with the results from the Wallington case study, where soil pH was found to cause significant variation with all other variables controlled.

5.3.2 Estimates using published soil survey data

5.3.2.1 NSRI

If no fieldwork time was available and SOC stocks for the Wimpole Estate were to be estimated knowing only the major soil group/soil series distribution on the estate and the mean %SOC values published in NSRI data for these soil series an r^2 value indicates that only 1.2% of the variation in %SOC on the Estate can be explained by the variability in major soil group %SOC from this database. This is likely the result of a large variation in other factors under the major soil groups/soil series, and the same inaccuracies discussed in relation to applying the mean %SOC values gained from fieldwork to the area of such soils. Although only 1.2% of the variation in %SOC on the Wimpole Estate can be predicted by the variation in major soil group %SOC values provided by NSRI data, this analysis does not reveal the difference between predicted and actual %SOC values. Interpolation of the NSRI predicted %SOC values would produce the %SOC map in Figure 5.10b. Comparison of Figure 5.10b with Figure 5.10a shows that the variation in major soil group can not explain the variation in %SOC, nor do the values ascribed to these major soil groups by the NSRI database accurately represent the actual %SOC values. An RMSE of 1.51 when the interpolated Estate actual %SOC values are plotted against the interpolated values predicted from the NSRI indicates this.

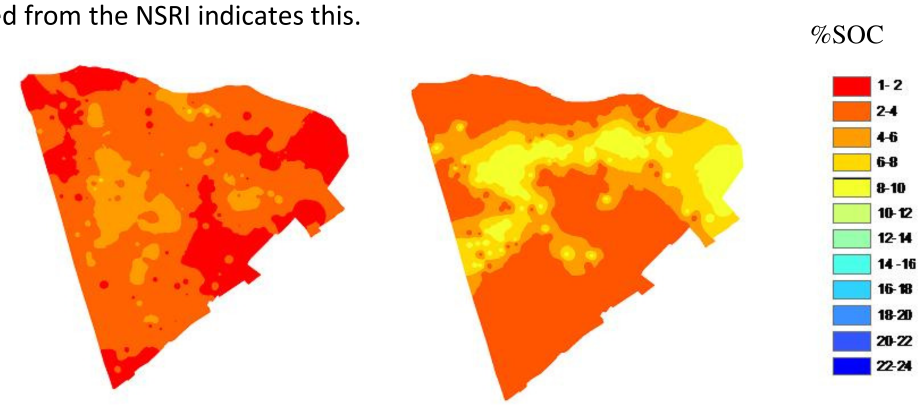

Figure 5.10a. Interpolated soil sample %SOC   b. interpolated NSRI soil series %SOC

The results of the paired t-test show that the use of NSRI soil series %SOC values is under-predicting the actual %SOC values by a mean of 0.56%. The t-value of -7.77 clarifies that there is a statistically significant difference in the mean of the actual %SOC values and the mean of those predicted using NSRI soil series data (Figure 5.11).

5.3.2.2 CSS data

Regression analysis of the 378 %SOC values from the Wimpole Estate against the mean values for major soil group from the CSS database indicates that only 1.22% of the variation in %SOC values could be explained by the variation in CSS mean %SOC's for major soil group. In addition to a lack of variation explained by the mean CSS %SOC's, an RMSE of 8.89 when interpolated CSS predictions are plotted against interpolated actual values reveals that the values ascribed to the major soil group by the CSS are too high for the Wimpole Estate (Figure 5.12).

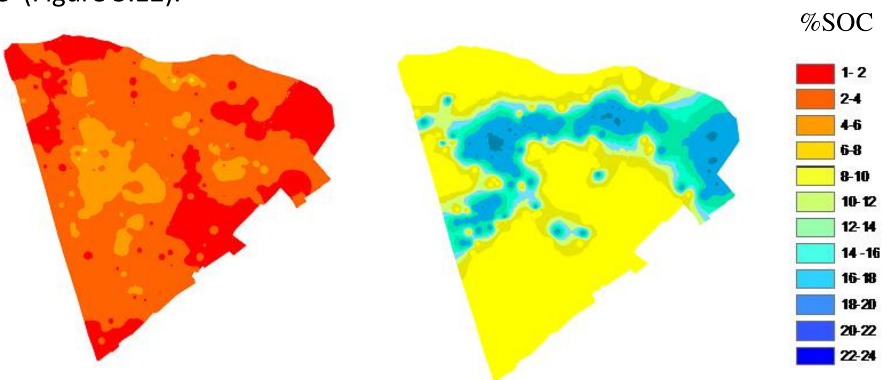

Figure 5.12a. Interpolated soil sample %SOC   b. Interpolated CSS major soil group %SOC

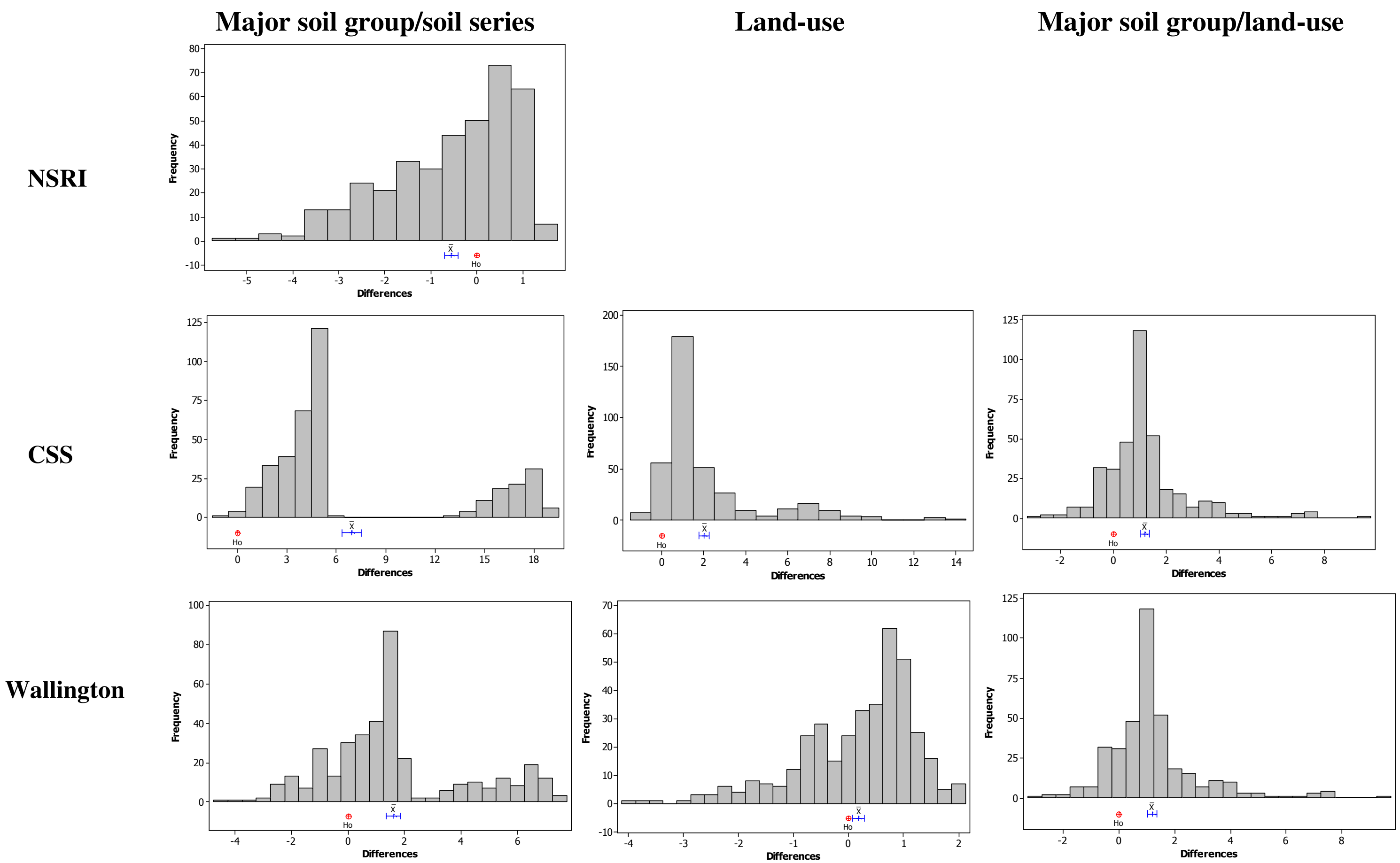

Figure 5.11: The differences (%) between soil sample %SOC values and %SOC values predicted using different databases and Wimpole Estate stratification schemes.

The results of the paired t-test show that the use of CSS major soil group %SOC values is over-predicting the actual %SOC values by a mean of 6.95%. The t-value of 23.37 ($p<0.05$) clarifies that there is a statistically significant difference in the mean of the actual %SOC values and the mean of those predicted using CSS major soil group data (Figure 5.11).

When mean values for land-use from the CSS database are regressed against actual %SOC 48.7% of the variation in %SOC can be explained by the variation in CSS %SOC mean land-use values. This indicates that if SOC estimates and maps are to be produced using either mean values for major soil group or land-use then land-use should be chosen. Although a greater amount of variation in %SOC can be explained by the variation in land-use the interpolated map of mean CSS land-use %SOC compared to the interpolated map of actual %SOC reveals that although more of the variation is being explained, the values ascribed to the land-uses are, as with major soil group (although not to the same extent), systematically too high (Figure 5.13). This is confirmed by an RMSE of 3.16.

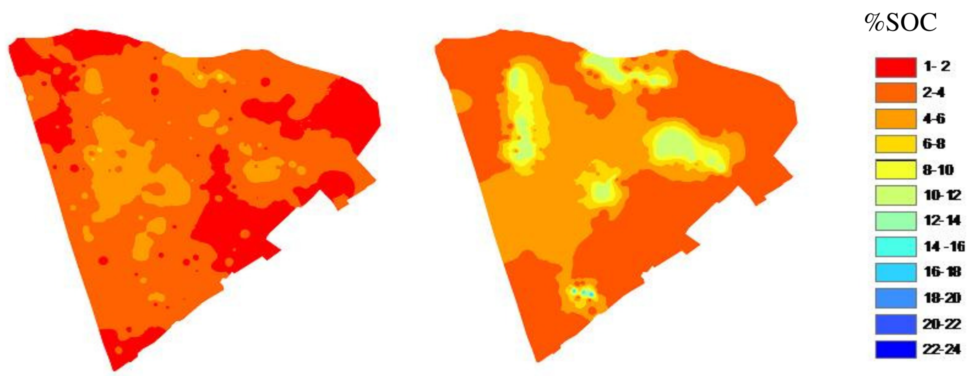

Figure 5.13a: Interpolated soil sample %SOC 5.13b: Interpolated CSS mean land-use %SOC

The results of the paired t-test show that the use of CSS land-use %SOC values is over-predicting the actual %SOC values by a mean of 2.04%. The t-value of 16.48 ($p<0.05$) clarifies that there is a statistically significant difference in the mean of the actual %SOC values and the mean of those predicted using CSS land-use data (Figure 5.11).

Using mean values from the CSS database for land-uses under a particular major soil group reveals that the variation in CSS major soil group/land-use %SOC values can explain 38.7 % of the variation in %SOC on the Wimpole Estate. This further emphasises the lack of %SOC variation explained by major soil group, and indicates that using the CSS values for land-use unadjusted for major soil group would provide the most accurate SOC stock estimate. An RMSE of 2.004 however reveals that the mean values assigned to land-uses under the individual major soil groups by the CSS are a more accurate indication of actual %SOC than simply using the mean values for land-use. This is emphasised by reference to Figure 5.14.

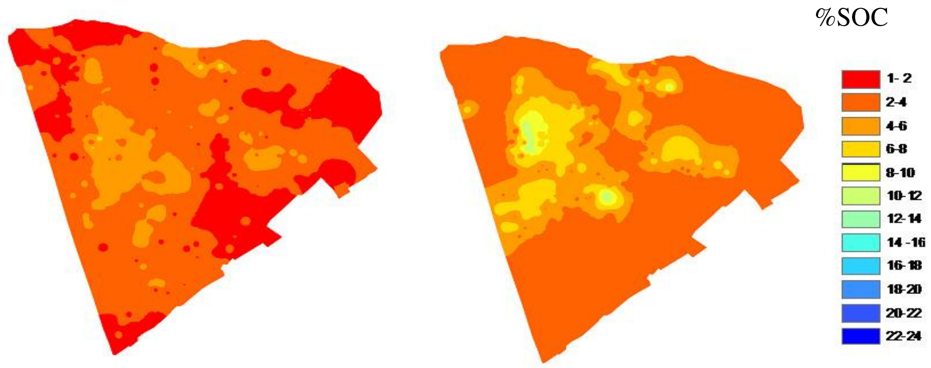

Figure 5.14 a. Interpolated soil sample %SOC b. Interpolated CSS mean land-use/major soil group %SOC

The results of the paired t-test show that the use of CSS land-use/major soil group %SOC values is over-predicting the actual %SOC values by a mean of 1.20%. The t-value of 14.51 ($p<0.05$) clarifies that there is a statistically significant difference in the mean of the actual %SOC values and the mean of those predicted using CSS land-use/major soil group data (Figure 5.11).

5.3.2.3 Wallington data

Regression analysis of the 378 %SOC values from the Wimpole Estate against the mean values for major soil group from the Wallington database indicates that only 1.22% of the variation in %SOC values can be explained by the variation in the Wallington mean %SOC values for that major soil group. An RMSE of 2.92 and reference to figure 5.15 indicates that although major soil group is not explaining the variation in %SOC, the %SOC values for major soil group from the Wallington database are more accurate than those from the CSS (Figure 5.12), however not as accurate as those from the NSRI (Figure 5.10).

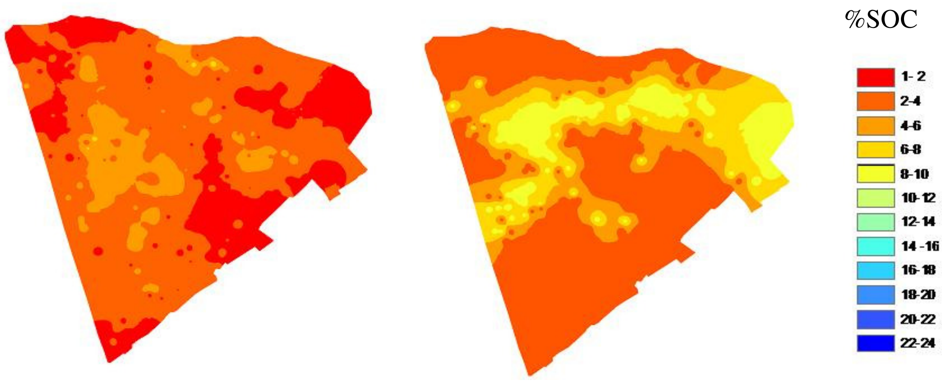

Figure 5.15 a. Interpolated soil sample %SOC b. Interpolated Wallington major soil group %SOC

The results of the paired t-test show that the use of Wallington major soil group %SOC values is over-predicting the actual %SOC values by a mean of 1.62%. The t-value of 12.52 ($p<0.05$) clarifies that there is a statistically significant difference in the mean of the actual %SOC values and the mean of those predicted using Wallington major soil group data (Figure 5.11).

Land-use %SOC values from the Wallington database can explain 59.23% of the variation in %SOC at Wimpole. This is a great improvement on using either database from the NSRI or CSS, and suggests that if no fieldwork is to be carried out then the %SOC values gathered from the Wallington study for land-use should be used to predict SOC stock estimates for other NT properties across the UK. Reference to Figure 5.16 and an RMSE of 1.07 shows that the land-use %SOC values from the Wallington database are a more accurate indicator of actual %SOC than the CSS land-use %SOC values or the NSRI soil series %SOC values.

Figure 5.16 a. Interpolated soil sample %SOC b. Interpolated Wallington land-use %SOC

The results of the paired t-test show that the Wallington land-use %SOC values over predict the actual %SOC values by a mean of 0.19%. The t-value of 3.45 ($p<0.05$) clarifies that there is a statistically significant difference in the mean of the actual %SOC values and the mean of those predicted using Wallington land-use data (Figure 5.11).

An ability to explain 58.83% of the variation in %SOC using mean values from the Wallington database for land-use on individual major soil groups, shows that using these values would not improve any %SOC estimates, further emphasising the insignificant role of major soil group in this study. Figure 5.17 and an RMSE of 1.79 shows that the mean values from Wallington for major soil group/land-use combinations, lead to a greater number of over-predictions than using mean values for land-use.

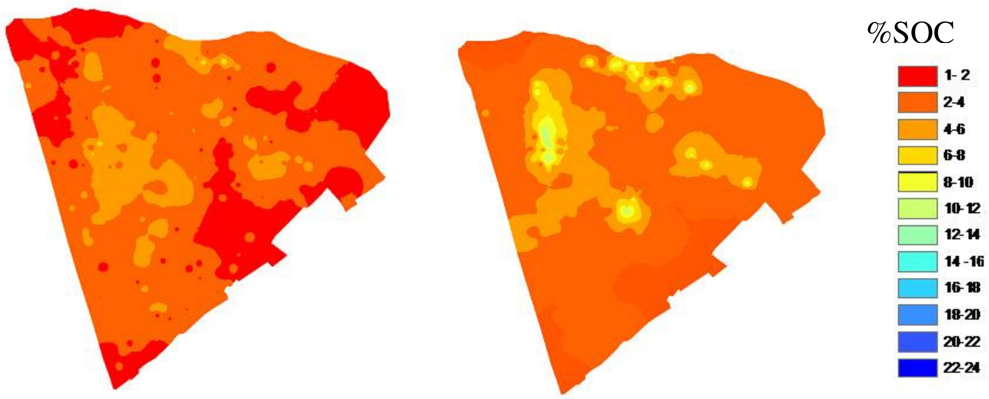

Figure 5.17 a. Interpolated soil sample %SOC b. Interpolated Wallington land-use/major soil group %SOC

The results of the paired t-test show that the Wallington land-use/major soil group combination %SOC values over predict the actual %SOC values by a mean of 0.73%. The t-value of 8.69 ($p<0.05$) clarifies that there is a statistically significant difference in the mean of the actual %SOC values and the mean of those predicted using Wallington land-use/major soil group combination data (Figure 5.11).

When the regression equation including altitude from the Wallington model was used to predict the %SOC values at Wimpole, only 20.5% of the variation could be explained by the variation in the predictions. The %SOC values predicted from this equation were generally too low, as seen in Figure 5.18. An RMSE of 1.72 confirms that including altitude in the prediction of SOC stock would lead to a great under-prediction.

Figure 5.18 a. Interpolated soil sample %SOC b. Interpolated Wallington land-use/major soil group/altitude %SOC

Results of the paired t-test show that the Wallington land-use/major soil group combination %SOC values, when adjusted for altitude, are under-predicting the actual %SOC values by a mean of 1.16%. The t-value of -18.13 ($p<0.05$) clarifies that there is a statistically significant difference in the mean of the actual %SOC values and the mean of those predicted using Wallington land-use/major soil group combination data adjusted for altitude (Figure 5.11).

The predictive errors of using each of the databases discussed, and the different methods of prediction are summarised in Table 5.1, indicating that the most accurate estimates would be achieved using average %SOC values from the Wallington database.

	Major soil group/series	*Land-use*	*Major soil group/land-use*	*Major soil group/land-use/altitude*
NSRI	*- 0.56*			
CSS	*+ 6.95*	*+ 2.04*	*+ 1.20*	
Wallington	*+ 1.62*	***+ 0.19***	*+ 0.73*	*- 1.16*

Table 5.1 The %SOC prediction errors (%) associated with the use of different databases and prediction methods. A positive error indicates an over-prediction and a negative error indicates an under-prediction. Cells highlighted in red indicate the most accurate method for each database and the number in bold indicates the most accurate of all estimates.

5.4 General discussion

Although the results of the Wallington study suggest that land-use history is not responsible for any of the variation in %SOC, it must be realised that the estate has not witnessed a large amount of land-use change in recent years (see Chapter 2). The fact that land-use history appears to be responsible for variations in %SOC on the Wimpole Estate is therefore not in contradiction to the Wallington findings, and suggests that any future SOC stock estimates on other estates should consider the variable land-use history as being responsible for variations in %SOC, in addition to those variables identified in Chapter 2. The results of this study suggest that arable land that has previously been in grassland use will have a higher %SOC than arable land that has been in continuous arable. This is to be expected due to reduced tillage under grassland and the greater amount of organic matter input. The results of this study agree with the work of Post and Kwon (2000) and Conant et al (2001), both of whom suggest greater SOC under arable land that has reverted to pasture land, however they are in conflict with Brye et al (2002) and Breuer et al (2006), where either no significant differences or a decrease in %SOC is found under pasture land compared to cropland. The findings of higher %SOC under arable land that was previously in pasture are supported by the finding of higher %SOC under pasture land that was previously cropland. These conflicting results from the literature suggest that more research is required into the impacts of land-use and land-use history on %SOC, and although land-use history was found to control %SOC at Wimpole, issues relating to the transition time for %SOC to reach equilibrium following land-use change, (Chapter 6) indicate that more research is needed into the detail of land-use history that should be included in calculations of SOC stocks.

Unlike in the Wallington study (Chapter 2) soil pH at Wimpole had an insignificant effect on %SOC when all other factors were controlled. The insignificant effect of pH in this study could be due to the lesser variation in soil types across the estate. The large variation in soil series present on the Wallington Estate, and the presence of a number of mineral soils with organic rich top-soils, may mean that some of the soil series at Wallington have been incorrectly mapped by the NSRI, and that the significance of soil pH in that study is due to the variation in soil types which has been obscured by the inaccurate mapping.

Again, unlike in the Wallington study the factor Farm was not found to be responsible for differences in %SOC. The fact that Farm was found to be an insignificant control on %SOC at Wimpole could be the result of all but one farm on the Estate being under arable land-use. Although there was variation between arable farms on the Wallington Estate, it was the variation under rough/semi-improved pasture which was most evident (see Chapter 2). The results of Chapter 2 and Chapter 3 suggest that land-management differences between farms in relation to their fertiliser treatment of pasture is the most likely reason for the %SOC variation. If, as expected, this is the case, the insignificant effect of Farm at Wimpole should be expected, as there is only one farm on which pasture management is undertaken, and therefore no comparisons can be made. It should thus not be concluded from this study that land-management differences between farms is not a relevant factor controlling %SOC, but instead that much more research is needed in this area. It neither should be concluded that arable land management techniques are an insignificant control on %SOC, as no detailed land-management questionnaire was undertaken for the Wimpole Estate, and it is possible that the methods of arable land-use often thought to be responsible for variations in %SOC (tillage, cover crops, rotation systems) were constant and that therefore land-management was not found to cause variation in %SOC simply because variation in land-management did not exist.

In order to try and resolve the issues surrounding fertiliser and stocking rates on pasture %SOC any further similar studies should focus on NT estates with large areas of grassland, owned and managed by a number of tenant farmers, and detailed information should be sought relating to the exact methods and types of land-management.

With the variables land-use history and farm entered into a GLM 62.30% of the variation in %SOC could be explained. Although this value is lower than the value of 66.65% which can be explained by other factors at Wallington, the lack of soil series data is likely to be responsible for much of this.

5.5 Conclusions

In this study land-use and altitude were found to be the most important controls on %SOC, with major soil group an insignificant variable. This does not however suggest that soil type is not responsible for variation in %SOC, as the scale of the soil map used could not identify individual soil series.

The farm tenant (and hence land-management) had no control on %SOC at Wimpole, however the majority of farms were in arable land-use, therefore land-management impacts on grassland remain unresolved. Past land-use was found to have a significant effect on %SOC, and should therefore be considered as a factor in any future SOC stock estimates.

If no fieldwork can be undertaken the most accurate SOC stock predictions for other NT estates will be achieved from knowing the land-use distribution on the estate, and by application of the mean %SOC values from the Wallington database to each respective land-use.

Chapter 6

Modelling the impacts of land-use change

A reformatted version of this chapter has been submitted to the journal *Global Biogeochemical Cycles*, and is currently in review. This paper is co-authored by Fred Worrall, Pete Smith, Anne Bhogal, Helaina Black, Allan Lilly, Declan Barraclough and Graham Merrington. I carried out all data processing, data analysis and model construction. I wrote the manuscript in its entirety and then passed it on to the co-authors for feedback. Fred Worrall provided support and guidance throughout, specifically in relation to model development and data relating to land-use change. He provided critical feedback and helped to guide the discussion of the manuscript. Pete Smith provided critical feedback and useful suggestions to guide the discussion. Anne Bhogal, Helaina Black, Allan Lilly, Declan Barraclough and Graham Merrington were involved in the provision of %SOC data.

6.1 Introduction

The UK Climate Change Act of 2008 calls for a reduction in GHG emissions of 80% by 2050, and a reduction in CO_2 emissions of at least 34% by 2020 (Ostle et al*.,* 2009). Although fossil fuel and agricultural emissions are major contributors to these high levels of atmospheric GHGs, the importance of carbon fluxes from soils (Chapter 1), and their contribution to either increasing or reducing atmospheric levels must not be overlooked. The fact that soils store 1550 Pg carbon globally in the top 100 cm (Lal, 2008) emphasises their importance in the global carbon cycle. The results of this thesis so far have indicated land-uses under which %SOC is expected to be high, and those under which it is expected to be low, as well as indicating the most appropriate methods to estimate SOC stock. Based on these findings the NT's aim to identify land-use interventions to increase carbon storage (Section 1.3) can be made, however the impact in terms of actual carbon emissions or sequestration can not be envisaged based on such information. For the impacts of land-use change on soil and atmospheric carbon to be fully understood requires not only knowledge of land-use change, but also the area of land to which land-use change will be made, and the time period over which the new land-use will remain in existence following land-use change. To allow the results to be useable by NT managers and land tenants, and meet the aim to ensure the results are transferrable to the entire NT landholding (Section 1.3) a soil carbon model was created, which forms the basis of discussion in this chapter.

6.1.1 SOC flux and land-use change: a review of the literature

Although several attempts have been made to estimate fluxes from SOC and establish the controls on this SOC store, there is still a lot of uncertainty over the exact extent of this store, and the processes by which gains/losses occur (Schulp et al., 2008). The general consensus is that land-use plays a major role (Scott et al., 2002), with soils acting either as a carbon sink or a carbon source as they approach a new equilibrium following land-use change (Guo and Gifford, 2002), but uncertainties in the magnitude of change remain. It is widely accepted and reported that SOC stocks differ under differing land-uses, and that carbon is released during conversion from grassland or forest to arable land, and accumulated following land-use change in the opposite direction (Howard et al., 1995; Zaehle et al., 2007; Post and Kwon, 2000; Veldkamp, 1994; Guo and Gifford, 2002). The extent of these changes is however uncertain, and a range of values have been reported for equilibrium SOC stocks under the land-uses in question. There are also large variations in reported transition times for SOC to reach a new equilibrium as a result of land-use change. Some studies assume linear transitions in soil carbon for all land-use changes over time periods of less than 20 years (e.g. Maia et al., 2009), whilst others have adjusted these to longer time periods (e.g. Tomlinson and Milne, 2006), and others assume instant change (e.g. Falloon et al., 2006). According to Powers (2004), there is a lack of studies looking at the effect on SOC of simultaneous land-use transitions.

Some research on stock changes in the soils of England and Wales suggests that SOC losses to the atmosphere could be increasing (e.g. Bellamy et al*.,* 2005), although how much of this is due to climate change is unclear (Smith et al*.,* 2007). These reported losses assume that land-use has remained constant over time, and are not due to land-use change. Although some studies suggest that SOC loss has increased where land-use has remained constant (Bellamy et al., 2005), these losses may have been counteracted by changes in land-use over this period in other areas, and therefore it may be incorrect to assume that because SOC loss has increased over time under land-use that has remained constant, that a greater percentage of the atmospheric GHG levels are made up of contributions from SOC emissions.

6.1.2 Chapter aims

The aim of this chapter was to produce a soil carbon model that would allow individual land tenants or owners to input future land-use change scenarios into the model and assess the impacts that these changes would have on levels of atmospheric carbon. The impact of simultaneous land-use change could be established, allowing identification of land-use change which could be made if other simultaneous change was made to offset these emissions. In addition to model creation this chapter also aims to use historical land-use change as a real-world example, to demonstrate potential contributions of land-use change to increasing atmospheric carbon levels.

Using UK land-use change over the period 1925 – 2007 as an example, the carbon emissions associated with historical land-use change decisions can be observed, allowing the impacts and consequences of similar future land-use change decisions to be identified. The research reported here aims to establish the role that changing land-use has had on SOC stocks and fluxes, and to therefore clarify the likely contribution that SOC has made to atmospheric GHG emissions over the last 80 years from the UK. The findings will help to quantify the impacts of land-use change in order to guide decisions on such change in the future. The approach taken by this study allows us to assess the role of land-use change on the emission or sequestration from, or to, SOC, and to investigate whether some land-use changes will increase SOC fluxes to the atmosphere, but can still be undertaken if a simultaneous land-use change will counteract this flux. The approach also allows investigation of the effects of using %SOC values collected from different datasets and different locations when applying these values to a large scale study. Most importantly, the model provides an estimate of the historical sequestration/emission of carbon from/to the atmosphere, and indicates the contribution of SOC change to the increasing trend in GHG levels.

The advantages of this current modelling study are that it utilises %SOC values taken from a large database, thereby increasing the range of conditions under which %SOC was measured, and it also takes into account the variation in transition times between %SOC values when land-use change occurs. The model is therefore an advance on the IPCC guidelines of using a single value of 20 years for duration of change (Smith, 2004), and should provide a better estimate as to how flux from SOC responds to land-use change.

6.2 Approach and Methodology

Howard et al. (1995) explain that future SOC stores following a change in land-use can be projected if you have a matrix of land-use change over time and a record of SOC stocks for particular land-uses. This approach is used in the current Land Use, Land-Use Change and Forestry (LULUCF) inventory for the UK and is also employed here. The LULUCF estimates of SOC change are however based on SOC stock values for the top 30cm of soil, as opposed to the top 20cm of soil in this study, making comparison of results difficult. This study considers how SOC stocks in the past have been affected by land-use change, and therefore allows us to assess the extent to which land-use change has contributed to the UK's GHG emissions over the last eight decades.

The modelling of historical fluxes of carbon from UK soils required the following information: typical SOC stocks for soils under all land-uses at equilibrium; the transition time over which SOC levels adjust to a new equilibrium following all land-use changes; the UK's land-use change history and direction of land-use change.

6.2.1 Soil carbon stocks by land-use

Calculation of SOC stocks required information on %SOC and the bulk density of soils under different land-uses, as well as the depth of soil to which the SOC stock was to be calculated. All values used throughout this study refer to SOC stock change in the top 20 cm of UK soils, as this is the depth of soil in which SOC is likely to respond to land-use change (Woomer et al., 2001; Cheng and Kimble, 2001; Kimble et al., 2001).

Several different land-use classification schemes exist for the land-uses found in the UK, depending on the database in which they originate. Although classified in different ways, it was considered that the majority of these land-uses are very similar in both character and %SOC, and could therefore be re-classified into a uniform system, so that land-use transition matrices could be constructed. The land-use classification system chosen in this study consisted of five categories: *Arable; Temporary grassland; Permanent grassland; Woodland and Urban*. These were chosen under the assumption that the majority of land-uses could be assigned to one of these categories without difficulty, and that they were representative of the majority of land-uses covered in the databases. Due to classification differences between databases, an element of subjectivity was involved in selecting which land-use classifications to include in the broad land-use categories used here.

A database of %SOCs, containing a total of 24,777 soil samples was used to establish typical %SOCs for soils under these land-uses in the UK. This large database was amalgamated from 15 individual databases, some covering all areas of the UK, and others specific to individual countries or regions. The databases used were: National Soils Institute Inventory (NSI Invent, Falloon, 2002), National Soils Institute Horizon data (NSI horizon, Falloon, 2003), National Soils Institute 1984 survey (NSI 1984, Loveland, 1990), National Soils Institute 2001 survey (NSI 2001, Bellamy et al., 2005), Countryside Survey 1978 (CSS 1978, Environment Agency*,* 2001), Countryside survey 1998 (CSS 1998, Haines-Young et al., 2000), Representative Soil Sampling Scheme (RSS, Webb et al*.,* 2001), Scottish Executive-estimating carbon in organic soils (ECOSSE, Smith et al., 2007), National Soils Inventory Scotland (NSIS, Lilly et al., 2009), Northern Ireland Inventory 2005 (NI 2005, CEH et al., 2007; Tomlinson and Milne, 2006), Northern Ireland Inventory 1995 (NI 1995, CEH et al., 2007; Tomlinson and Milne, 2006), Wallington (Bell and Worrall, 2009), Wimpole (Chapter 5) and English Nature Woodland data (EW Wood, Chambers et al., 1998). Median and inter-quartile ranges were calculated for both the amalgamated and individual databases for all land-uses under consideration.

All databases included the category *Arable* and selection for this classification was straightforward. Eight of the databases did not include any soil samples from a land-use similar or representative of temporary pasture, therefore these databases were not included in this category, and for all model runs using individual databases in which this occurred, the median, and inter-quartile ranges from the amalgamated database were used. The same was true when any databases did not include any soils representative of permanent pasture or woodland.

The %SOC of urban land is debateable and is often assumed to be zero if the soil is removed during urban land conversion, and that built up areas contain no soil (Tomlinson and Milne, 2006). Others argue, however, that it will approximate the value of the land-use from which it was transformed (Howard et al., 1995). As none of the databases used here contained samples taken from a land-use representative of urban land it was assumed that a %SOC of 0% could be applied, based on the idea that all top-soil would be removed during ground preparation and foundation establishment. It is recognised however, that this value could vary depending on the exact use of urban land.

Data on soil bulk density was obtained in the same way as that for %SOC, by amalgamation of all databases, with a median value established for each land-use in question.

The amalgamated soil carbon database used in this study includes soil samples taken from all over the UK, and the values ascribed to each land-use class therefore cover a very large range of climatic, altitudinal and soil textural conditions - all factors which can result in differing SOC stocks (Krishnan et al., 2007; Yang et al., 2008). Due to a lack of knowledge concerning exact locations associated with the land-use change in question, however, it was concluded that using these UK-wide values was the best approach to take.

Although it is realised that organic soils and mineral soils have very different SOC stocks and respond differently to land-use change it was not possible to know exactly which land-use changes occurred on which soil types. The approach of using median values for land-uses from the UK-wide database means however that these different SOC values and behaviours following land-use change will be accounted for. A land-use change from pasture to woodland on mineral soils for example would likely result in sequestration of atmospheric carbon to SOC, whereas the same land-use change on organic soils could in-fact cause a release of SOC to the atmosphere. In this study the use of SOC values taken from a range of both mineral and organic soils means that the median and inter-quartile range values will have accounted for, for example, both these higher and lower SOC values under woodland compared to pasture.

A median %SOC value and inter-quartile range was calculated for all land-uses, for the amalgamated database and for each database individually.

6.2.2 UK land use history

The change in area of land-use for the UK over the period 1925 to 2007 was reconstructed using data from several sources. The initial year of 1925 was used as this was the year of formation of the UK within its current borders. Land-use information (area of arable crops, temporary grassland, permanent grassland, bare fallow, rough grazing, common rough grazing, set-aside and urban) was available for the UK from the Ministry of Agriculture, Fisheries and Food, and the Department of Environment, Food and Rural Affairs (MAFF, 1926 to 2000, DEFRA, 2001 to 2008). Information on the area of woodland over this period was available from the Forestry Commission (Forestry Commission, 2007). This woodland data was however only available for the years 1924, 1947, 1965, 1980, 1990, 1998-2002 and 2008. To get an annual estimate of woodland area linear interpolation was used between survey dates.

The land-use data provided by these sources was, as was the case with the SOC databases, classified differently to those chosen in this study. In order to fit the categories *Arable*, *Temporary pasture*, *Permanent pasture*, *Woodland* and *Urban* the following land-use categories from the original databases were grouped as follows: arable crops, bare fallow, set aside = *Arable*; temporary grass = *Temporary pasture*; permanent grass, rough grazing, common rough grazing = *Permanent pasture*; woodland = *Woodland*; urban = *Urban*.

Although information on land-use change covering the period 1925 to 2007 was available, this only provided detail on the change in area of each individual land-use, and did not reveal the direction of land-use change from and to another land-use. Expert judgement was used to allocate which land-uses were likely to convert to other land-uses. In some situations the direction of land-use change could confidently be estimated simply by observing a simultaneous increase/decrease of similar magnitude in two land-uses, therefore reaching the consensus that the land-use with a decrease in area was likely to have lost land to the land-use with an increase in area. In all situations the work of Adger and Subak (1996), and Adger et al. (1991) was used to help identify the most likely direction of land-use change. Assessment of a land-use change matrix shown in Adger et al. (1991) suggested that a change out of arable land-use would most likely result in conversion to land-uses in the following order: 1. improved pasture, 2. urban land, 3. permanent/rough grazing and 4. woodland. Adger and Subak (1996) supported a preferential loss from arable land to urban land over permanent/rough pasture and woodland. The land-use change matrix suggested a preferential loss from temporary pasture to 1. arable, 2. urban land, 3. permanent/rough pasture, 4. woodland, further supported by Adger and Subak (1996) and a loss to 1. arable, 2. urban land, 3. woodland. Both Adger et al.(1991) and Adger and Subak (1996) suggested that permanent/rough pasture would preferentially be lost to woodland, and Adger et al. (1991) suggested a loss then to 2. improved pasture, 3. arable, 4. urban land. A change in woodland was most likely to result from a change in 1. improved or permanent/rough pasture, 2. arable, 3. urban land, and a change in urban land was most likely to result from a change in 1. improved pasture, 2. arable, 3. permanent/rough pasture 4. woodland. The predicted directions of land-use change used in this study can be seen in Figures 6.1- 6.5.

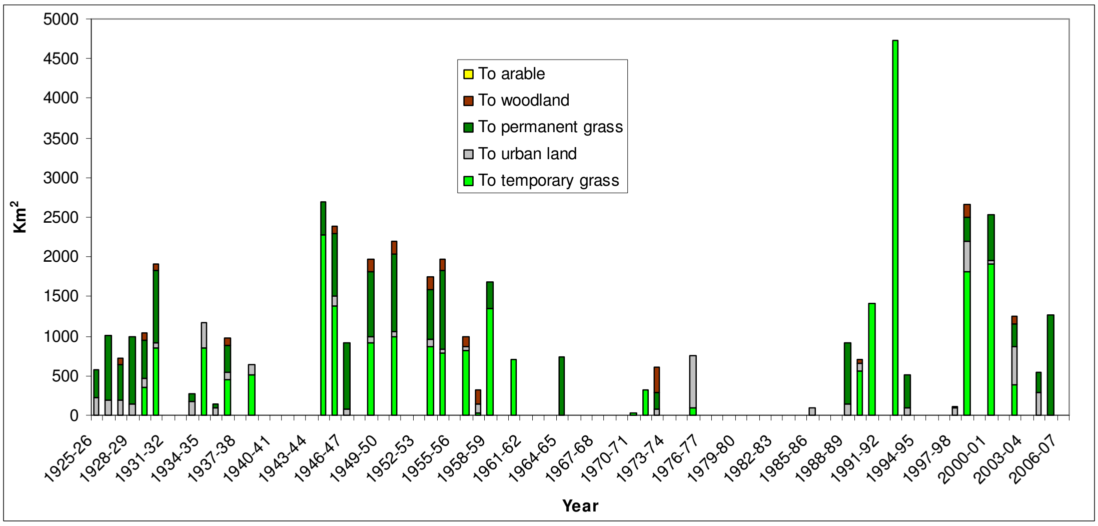

Figure 6.1 The predicted direction of UK land-use change out of arable, 1925-2007

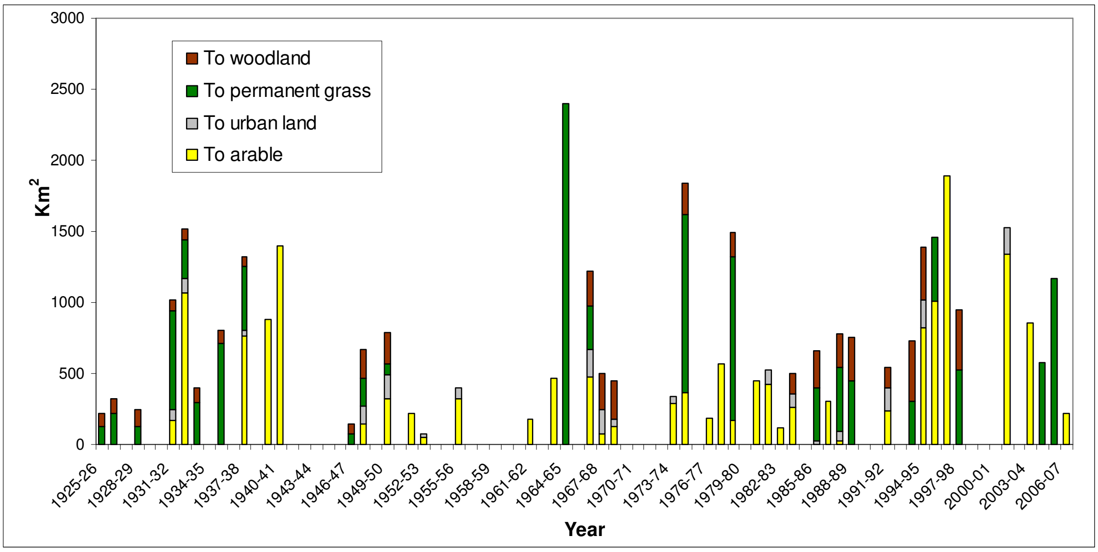

Figure 6.2 The predicted direction of UK land-use change out of temporary pasture, 1925-2007

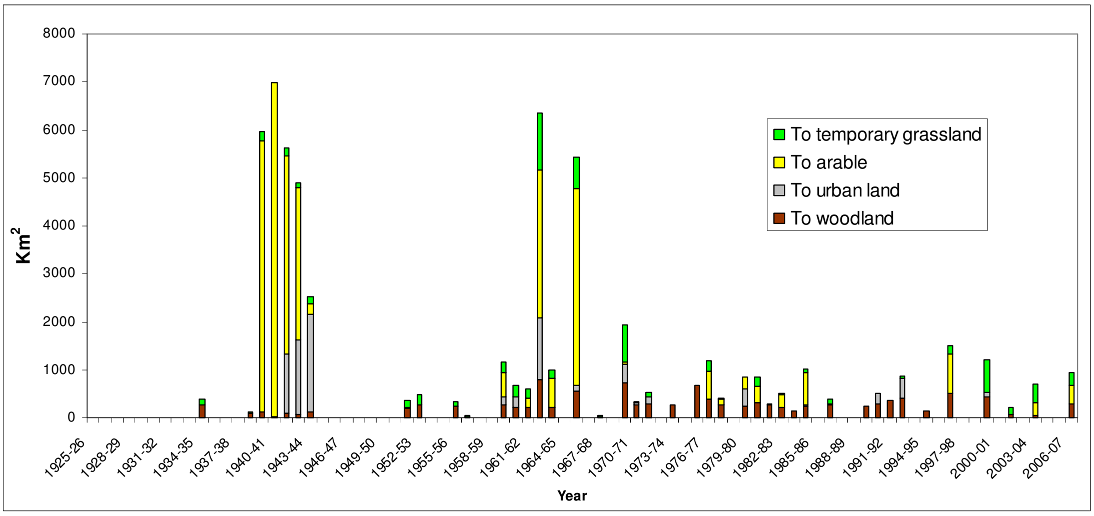

Figure 6.3 The predicted direction of UK land-use out of permanent pasture 1925-2007

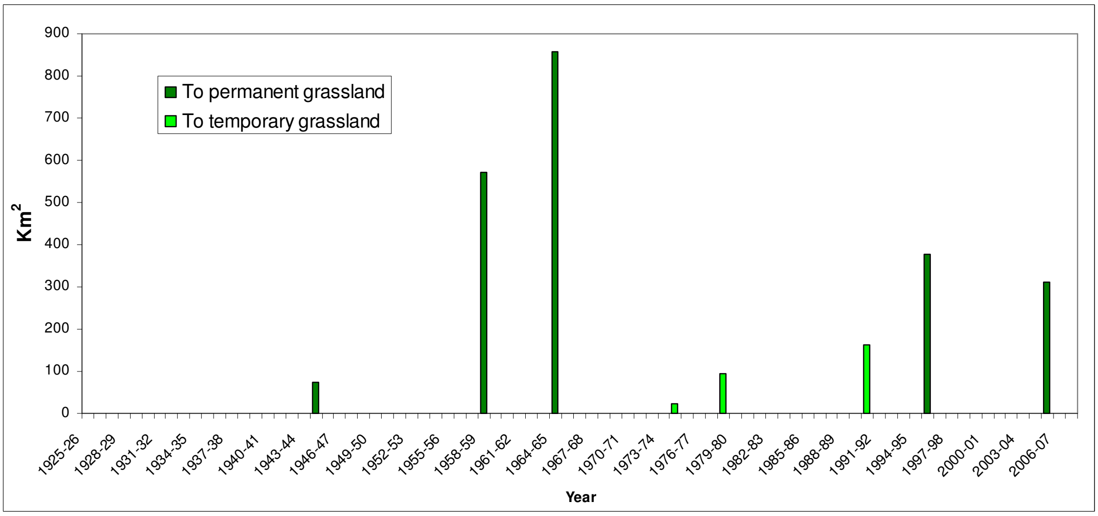

Figure 6.4 The predicted direction of UK land-use change out of urban land, 1925-2007

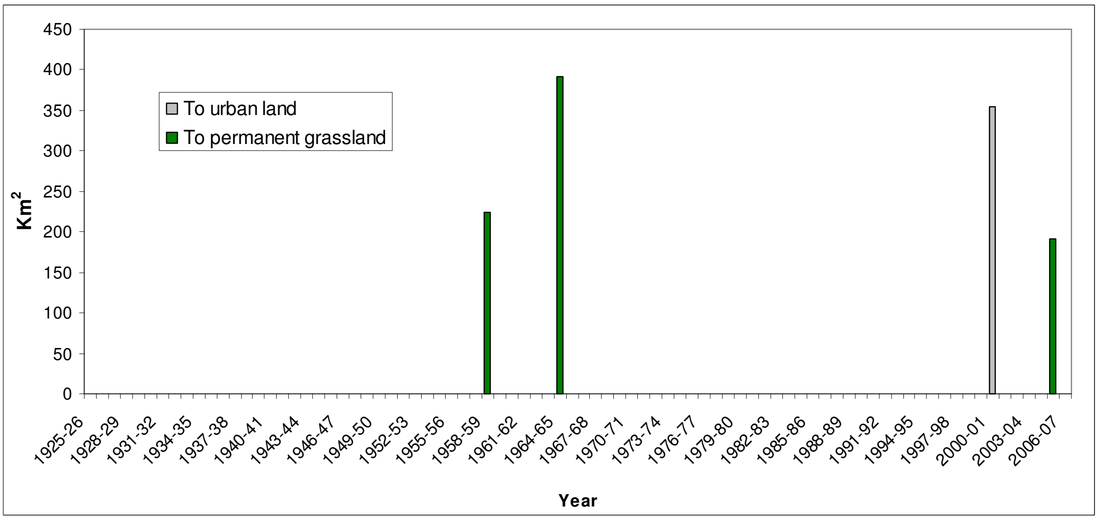

Figure 6.5 The predicted direction of UK land-use change out of woodland, 1925-2007

6.2.3 SOC transition times

A lack of long-term field trials measuring SOC change on a continual basis following land-use change means that there is still a large amount of uncertainty over both the transition time and the rate of transition as soils approach a new equilibrium. There is no consensus as to when SOC equilibrium will be reached following land-use change- with estimates ranging from 6 years to 100 years (King et al., 2004; Smith et al., 1997).

To achieve a best approximation of these transition times it was considered that an in-depth literature review would reveal the most realistic and reliable outcome. Although many studies were qualitative rather than quantitative in their reference to transition times, the information provided by each study was used to establish decay constants, with the reasoning behind these calculations outlined in Tables 6.1 and 6.2.

Land-use change	Reference	Time taken	Rate of change	Estimated decay constant (/yr)
W to A	Mann, 1986	20 years +	Fastest in first 20 years then gradual.	
W to A	Murty et al, 2002	75 years approx 18.75 half lives	Initially fast, then much slower. 75 years: 72% ↓ from woodland 3 years: 31% ↓ from woodland	0.170
W to A	Bonde et al, 1992	50 years approx 7.94 half lives	50 years = 59% loss 12 years = 56% loss	0.110
W to A	Motavalli et al, 2000	20 years approx 7.04 half lives	5 years = 44% loss. 50% loss reached steadily over next 20 years.	0.244
W to A	Houghton and Hackler, 2000	15 years		0.520
W to A	Woodbury et al, 2006	Many years	Exponential	
W to A	West et al ,2004	20 years	Exponential	0.390
W to A	Murty et al, 2002	41 years	Exponential	0.190
W to A	Heath et al, 2002	10-15 years	Rapid	0.620
W to A	Heath et al, 2002	20-50 years	Rapid for 20 years, then gradual.	0.220
W to A	Milne, 1999	50-150 years	Fast	0.080
W to A	Houghton et al, 1983		Initially rapid then slow	
W to A	Schlesinger, 1986	Over 60 years	Exponential- most rapid in first 20 years	0.130
W to A	Lubowski, et al, 2005	Immediate drop		0.693
W to P	Heath et al, 2002	25 years	Constant, linear	0.055
W to P	Milne, 1999	50-150 years	Fast	0.014
W to P	Lubowski et al 2005	Immediate drop		0.693
P to A	Milne, 1999	50-150 years	Fast	0.014
P to A	Climate leaders greenhouse gas inventory, 2008	5 years	Constant, linear	0.277
P to A	Lubowski et al 2005	Instant		0.693

Table 6.1 Literature review and reasoning for estimated decay constants associated with land-use change resulting in soil carbon losses, where W: Woodland; A: Arable; P: Pasture

Land-use change	Reference	Time taken	Rate of change	Estimated decay constant (/yr)
A to W	Houghton and Hackler, 2000	150 years	Rapid for the first 50 years, slower for the next 100 years	0.050
A to W	Post and Kwon, 2000	50-100 years	Constant, linear	0.019
A to W	Poulton et al , 2003	120 years	Constant, linear	0.012
A to W	Faloon et al, 2006		Gradual	
A to W	Falloon et al, 2004	100 years	Linear	0.014
A to W	Milne, 1999	100-300 years	Slow	0.007
A to PW	Falloon et al , 2006	Instant		0.693
A to PW	Woodbury et al, 2006	Many decades	Exponential	
A to PW	Grogan and Matthews, 2001	102 years	Linear	0.014
A to PW	Andress, 2002	125 years	Fast, becoming slower	0.017
A to PW	Hansen, 1993		Initial loss for 6-12 years, gain after 18 years	
A to PW	Heath et al, 2002	50 years	Slow and gradual	0.028
A to PW	Heath et al, 2002	10-200 years	Immediate and constant/linear	0.015
A to PW	Zak et al, 1990 (Cited in Heath et al, 2002)	60 years	Initial loss for 10 years. Gradual and constant linear gain over next 50 years	0.028
A to PW	Grigal and Berguson, 1998		Initial loss for 5 years, recovers to original level by year 15	
A to PW	Paul et al, 2002		Initial loss for 5 years, recovers to original level after 30 years, then gains	
P to W	Paul et al 2002		Initial loss for 5 years, recovers to initial level after 30 years then gains	
A to P	Freibauer at al, 2004	20-100 years	Non-linear: Initial gain then slows. 50% gain after 47 years	0.015
A to P	Climate leaders greenhouse gas inventory, 2008	20 years	Constant, linear	0.069
A to P	Lubowski et al, 2005	40 years	Exponential	0.038
A to P	Lee et al, 2005	30 years	Linear	0.046
A to P	Falloon et al, 2004	50 years	Linear	0.028
A to P	Jenkinson et al, 1987	35 years	Linear	0.040
A to P	Milne, 1999	100- 300 years	Slow	0.007

Table 6.2 Literature review and reasoning for estimated decay constants associated with land-use change resulting in soil carbon gains, where: W: natural woodland; PW: planted woodland; A: arable; P: pasture

The decay constants from each study were combined to obtain a median decay constant for each land-use change transition. The lack of information in previous studies relating to transition times meant that when transition times were referred to it was very unlikely that they were specific to temporary and permanent pasture, with the majority of cases only referring to the land-use as pasture. All land-use transitions into or out of both temporary and permanent pasture were therefore ascribed the same values. In the case of a land-use change into urban land it was considered that the change would occur immediately, as the soil would be removed from the site and all SOC would disappear instantly. In terms of transitions out of urban and into arable or pasture it was assumed that the transition would occur at a similar rate to that of a transition from arable to pasture, due to a similar extent of SOC change associated with such a land-use change. Although a large number of studies looked at transition times associated with arable to woodland, there was a lack of studies looking at any other change from woodland. It was therefore assumed that these transitions would occur at similar rates.

6.2.4 SOC flux model

The assumption was made that all SOC transitions followed first order rate kinetics. Initially the model does not make any account for climate change over the period, to enable the extent of land-use change contributions to SOC emissions, and hence to GHG emissions to be established. An adjustment for climate change was not considered appropriate at this stage as the databases from which the SOC values have been amalgamated span a period of several decades (1970-2009), and therefore the median values ascribed to these land-uses are likely to have already accounted for some of this change resulting from climate.

The approach taken here considers each transition between any combination of land-uses as a first-order kinetic process, and then that the flux from the soils is the inter-annual change in the soil carbon stock. The SOC stock for each year was calculated using Equation 6.1 and Equation 6.2.

S_{dt} = K $$S_{dt} = K\sum_{1}^{t} \sum_{1}^{j} \sum_{1}^{i} A_j d(C_i \rho_{bI} - C_j \rho_{BJ}) e^{-\lambda t}$$

Equation 6.1

$$F_t = S_{dt-1} - S_{dt}$$ **Equation 6.2**

Where: F_t = the flux of carbon from the UK soils in year t (tonnes C); S_{dt} = the carbon stock in UK soil to depth d in year t (tonnes C); d = the depth of the soil layer considered (m); A_j = the area of land use j that transitions to land use i (m^2); C_x = the organic carbon content of the soil in land uses i and j (%). ρ_{bx} = the bulk density of land uses i and j (kg m^{-3}); λ= the time constant for transition between land uses i and j (yr^{-1}); and K = conversation factor for equalising units. Note that this equation is written such that negative flux (F_t) is equivalent to carbon loss from the atmosphere and addition to soil.

The model was run stochastically with 100 values drawn at random based upon a uniform distribution from the ranges obtained for %SOC, the median values for bulk density and the median value for land-use transition decay constants. The model was run and an output generated using both the amalgamated database and several of the regional and country specific databases in order to compare outputs generated using different SOC values.

6.2.5 Climate effect

The model outlined in Equations 6.1 and 6.2 does not take climate change into account. An additional model was therefore run, utilising the approach of Smith et al (2007) (Equation 6.3) to allow establishment of the total contribution of SOC to UK GHG emissions.

$$Q_{10} = \left(\frac{R_2}{R_1}\right)^{\left(\frac{10}{T_2 - T_1}\right)}$$ **Equation 6.3**

Where: Q_{10} = the rise in reaction rate for a 10 K rise in temperature; R_x = soil respiration at temperature T_x; and T_x = the air temperature at x (K).

For the UK the respiration rate is considered to be in equilibrium with the mean annual temperature as represented by the Central England Record (www/metoffice.gov.uk). The approach of Smith et al, (2007) suggested that Q_{10} =2 and that the reaction rate was 0.01 yr^{-1}, giving a respiration rate for the UK of between 1.3 and 2.6 Tg C/yr. In order to include climate change in the model it was necessary to assume a point in time for the SOC values to be in equilibrium with the respiration rates proposed by Smith et al. (2007). The question is then at what point in time are the SOC value ranges used in this study valid. This study uses three scenarios for assessing the impact of climate warming over and above the influence of land-use change:

1. The SOC and respiration data is true for the year before the study starts, i.e. 1924.
2. The SOC and respiration data is true for the year at the end of the study, i.e. 2007.
3. The SOC and respiration data is true for median year of the soil survey data, i.e. 1987.

Each scenario was considered using the extremes of the respiration rate proposed above, i.e. six models runs were performed. The upper, lower and median of these six runs were selected on the basis of the sum of carbon losses over the entire study period, these three were then added to the upper, lower and median estimates as predicted from land use change alone to give a combined land use/climate change estimate of release from SOC.

6.3 Results

6.3.1 UK land use change 1925-2007

The percentage of land under both permanent and temporary pasture is lower in 2007 than in 1925 (Figure 6.6).

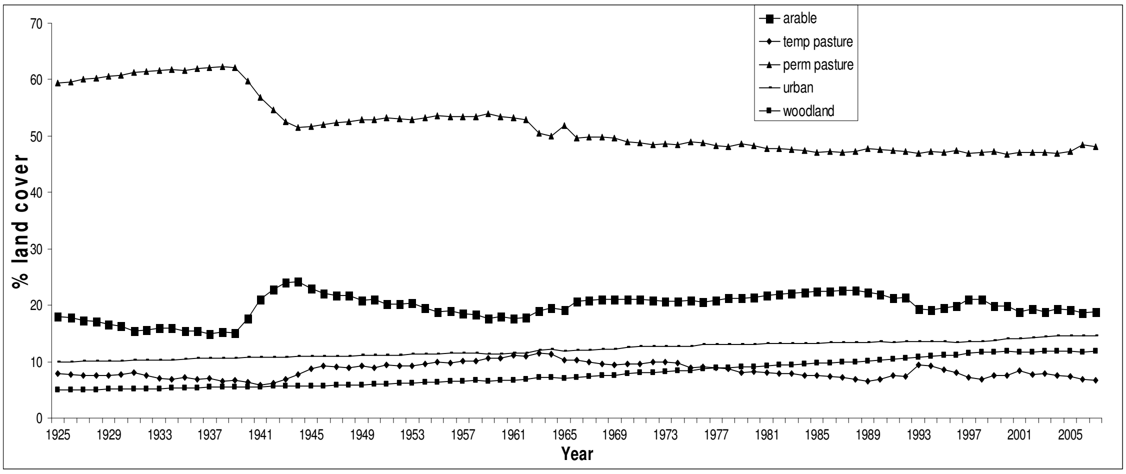

Figure 6.6 UK land-use change 1925- 2007.

There was a slight increase in permanent pasture until 1939, followed by a sharp decrease into the early 1940s, and a steady decline thereafter. Permanent pasture at the beginning of this time period covered approximately 59% of the UK land area, and now covers less than 49%. The area under temporary pasture has fluctuated over the period, and has covered a significantly smaller area than permanent pasture throughout. The land area under arable crops has been second only to permanent pasture throughout the period, and has generally followed a reverse trend to that of permanent pasture, with a large increase in area in 1939, followed by several fluctuations in reverse to that of either permanent pasture or temporary pasture. Other notable fluctuations in arable land area were an increase in the early 1960's and a decrease in 1991. Land area under urban has increased very steadily over the period, retaining its position as the third greatest land-cover. Since 1925 it has gained approximately 50% of its original land area- increasing its percentage cover from 10% to approximately 15% by 2007. Land area under woodland increased most rapidly in the early 1960's and then at a steady and constant rate thereafter to replace temporary pasture as the fourth most important land cover, resulting in a more than 50% increase in its percentage cover from 5% to greater than 10%.

6.3.2 %SOCs by land-use

There was an increase in %SOCs with land-use in the order: *Urban*<*Arable*< *Temporary pasture* < *Permanent pasture* < *Woodland* (Table 6.3 and Table 6.4). SOC values for the same land-use did however differ between databases (Table 6.3). This is to be expected due to some databases being regionally specific (e.g. Wallington) and others being specific to countries with large areas of organic and peat soils (e.g. NI and NSIS). It should also be noted that given the ranges of observed values it is possible that some land-use changes that would most commonly result in a SOC decrease may on occasion result in an increase and vice versa.

Database		All	RSS	NSI 1984	NSI 2001	NSI Invent	NSI horizon	Ecosse	CSS 1978	CSS 1998	Wallington	NI 2005	NI1995	NSIS	Wimpole	EW wood 2001	EW wood 1971
		(24777)	(9961)	(5121)	(2143)	(1543)	(1136)	(973)	(867)	(841)	(598)	(484)	(457)	(312)	(282)	(90)	(90)
Arable	median	2.17	1.94	2.20	2.06	2.06	2.30	4.02	2.50	2.47	2.73	3.88	4.17	3.31	2.01		
	Lower Q	1.50	1.37	1.50	1.57	1.59	1.70	3.00	2.00	1.94	2.28	2.44	2.42	2.23	1.71		
	Upper Q	3.07	2.62	3.30	2.73	2.74	3.10	5.12	3.50	3.36	3.20	5.06	5.48	4.18	2.42		
Temp grass	median	2.74	2.57	3.20	3.04	3.30	2.70				2.93			4.14			
	Lower Q	2.05	1.94	2.30	2.29	2.36	2.00				2.30			3.11			
	Upper Q	3.76	3.36	4.60	3.88	5.53	3.60				3.26			5.68			
Perm grass	median	4.40	4.22	4.70	4.17	3.22	4.00	5.42	4.50	5.05	3.87	6.15	5.20	7.40	3.64		
	Lower Q	3.24	3.14	3.30	3.08	2.34	3.00	4.40	3.00	3.66	3.13	4.38	3.90	4.65	3.10		
	Upper Q	6.02	5.36	7.40	6.06	4.40	5.50	6.52	8.00	7.88	4.93	11.0	8.31	2.71	4.38		
Woodland	median	7.00		4.80	5.10	5.94		30.5	7.00	9.60	27.2	50.5	42.1		4.73	9.12	8.21
	Lower Q	4.20		2.93	3.32	4.24		18.55	5.70	5.70	16.71	10.78	30.30		4.15	6.80	6.44
	Upper Q	17.63		7.70	7.75	10.0		40.48	27.68	27.68	33.74	53.70	46.03		5.79	11.54	11.09

Table 6.3 Databases used to predict SOC values under different land-uses. All = all databases combined, RSS = Representative Soil Sampling Scheme, NSI = National Soils Institute, NSI Invent = National Soils Institute Inventory, NSI horizon = National Soils Institute Horizon Ecosse = Scottish Executive: Estimating Carbon in Organic Soils, CSS = Countryside Survey, NI = Northern Ireland, NSIS = National Soils Institute of Scotland, EW = English Nature woodland data, Lower Q = lower quartile, Upper Q = upper quartile, Temp = temporary, Perm = permanent

	% SOC	bulkdensity (g/cm^3)	depth (cm)
Arable	1.50-3.07	1.22	20
Temporary grassland	2.05-3.76	1.22	20
Permanent grassland	3.24-6.02	1.02	20
Urban	0	1.22	20
Woodland	4.20-17.63	0.58	20

Table 6.4 %SOC, Soil bulk density and soil depth values used in model.

6.3.3 SOC transition times

The decay constants used in the model are shown in Table 6.5, indicating the variation in transition times depending on the land-use change in question. It can be seen that although many literature values would suggest that transition times are always simply reversible for all land-use changes, this is not the case, with SOC gains occurring at a slower rate than SOC losses. The transition times used in this model differ greatly from the 20 year linear change assumed (as a global simplification) by the IPCC (see Smith, 2004).

	Land-use b				
Land-use a	Arable	Temp	Perm	urban	Woodland
Arable	0.00	0.04	0.04	0.69	0.02
Temp	0.28	0.00	0.03	0.69	0.02
Perm	0.28	0.33	0.00	0.69	0.02
Urban	0.03	0.04	0.04	0.00	0.02
Woodland	0.06	0.06	0.06	0.69	0.00

Table 6.5 SOC decay constants with a change from land-use a to land-use b.

6.3.4 SOC flux

The model predicts a 5% net gain in SOC stocks between 1925 and 2007 due to land-use change (Figure 6.7), representing a gain of 102 Tg C. This equates to a median flux into the soil of 1.92 Tg C/year (inter-quartile range: 0.19 – 3.12 Tg C/year.)

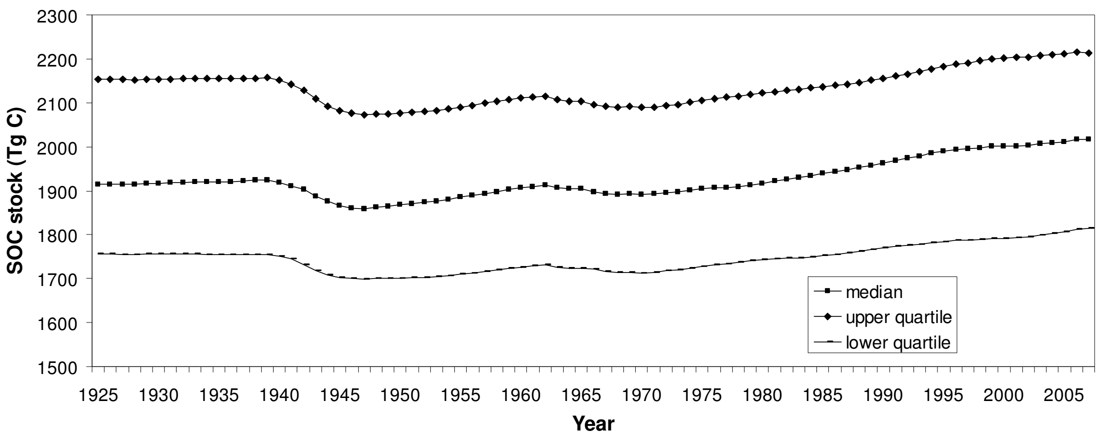

Figure 6.7 Modelled change in UK SOC stock: 1925- 2007

The greatest sink (loss from the atmosphere) in any one year occurred in 1993, with a flux into the soil from the atmosphere of 4.31 Tg C. The greatest loss of SOC to the atmosphere occurred in 1942, with a flux to the atmosphere of 12.30 Tg C. Other noticeable periods of carbon sequestration to the soil were during the entire decade of the 1950's, and for at least 20 years from 1970 into the early 1990's. Other than the emissions of 1942 there were also large fluxes of SOC to the atmosphere in 1961, 1964 and 2005. Examination of Figure 6.8 reveals that from 1970 to 1994 the UK's SOC flux remained relatively constant, sinking an average of 3.10 Tg C/yr, representing a net gain in SOC stock of 0.20% per year. This trend, however, may be beginning to decline, with a gradual decrease in the extent of the carbon sink appearing from 1994 onwards, with an average sink of only 2.00 Tg C/yr, a gain of only 0.12% per year.

The extent of the flux caused by transitions into and out of various land-uses over the entire period is compared in Figure 6.10. The change in SOC stock over the entire period is shown in Figure 6.7, and the timing and direction of yearly fluxes can be seen in Figure 6.8. The cumulative SOC flux is shown in Figure 6.9.

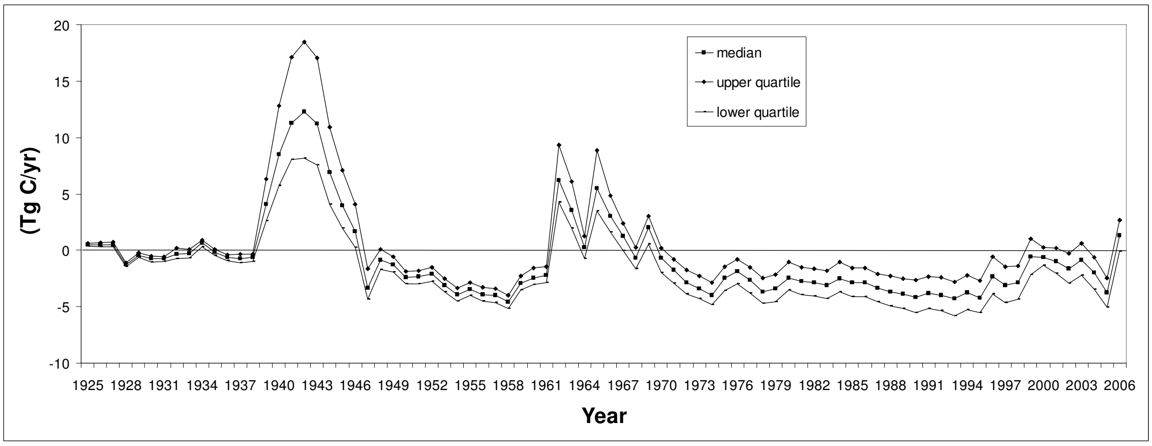

Figure 6.8 The modelled flux of UK SOC from 1925- 2007 resulting from land-use change. A positive flux represents a flux from the soil to the atmosphere; a negative flux represents a flux from the atmosphere into the soil.

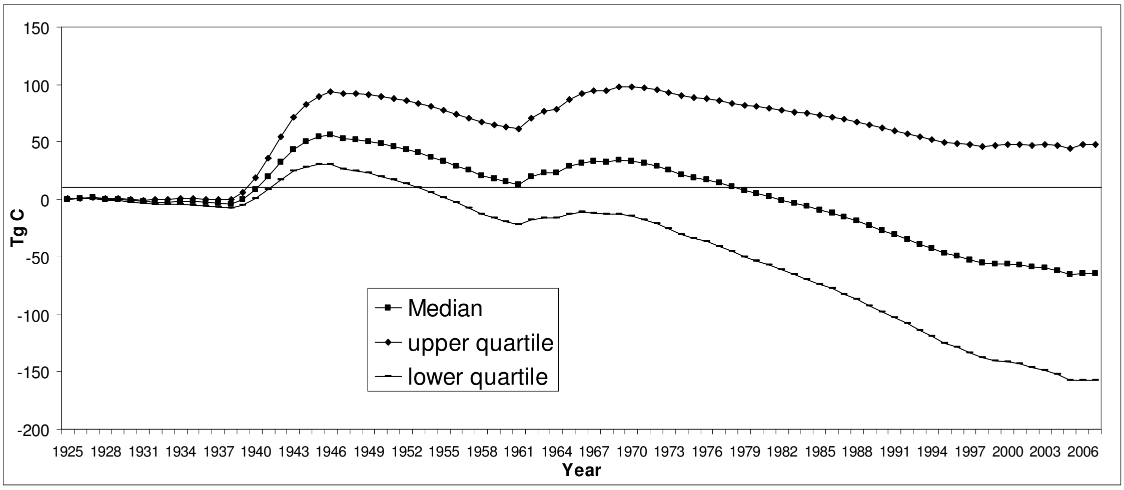

Figure 6.9 The UK's cumulative SOC flux: 1925-2007

Predicted SOC fluxes differed greatly depending on the database used (Figure 6.11), with the greatest total flux over the period predicted from the Wallington database, where a net gain in SOC content of 13.81% is compared to only 2.74% using the NSI Inventory values.

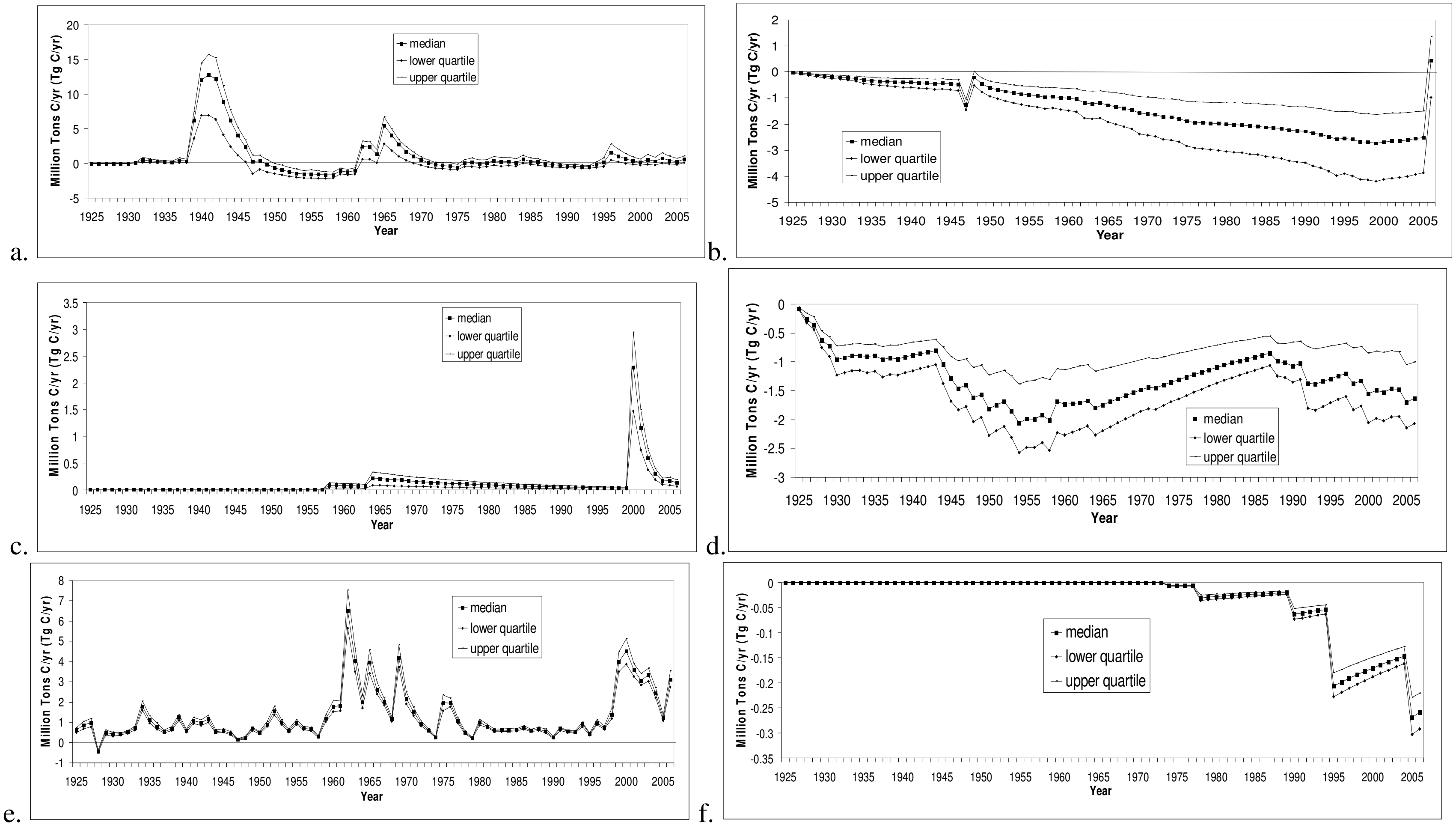

Figure 6.10 The extent and direction of the overall SOC flux caused by various land-use change transitions. a. Land-use change from grassland to arable; b. Land-use change into woodland; c. Land-use change out of woodland; d. Land-use change from arable to grassland; e. Land-use change into urban f. Land-use change out of urban

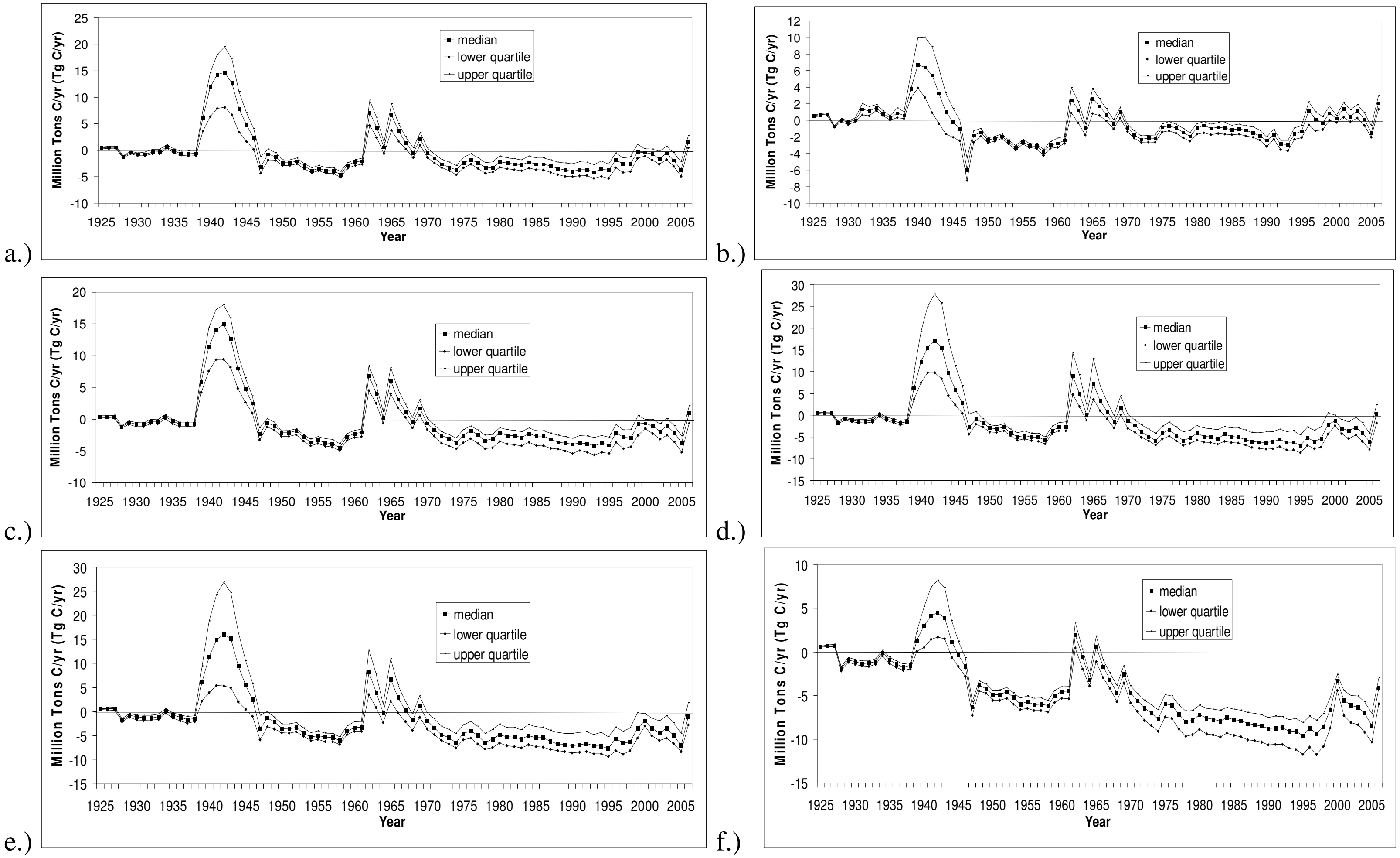

Figure 6.11 The difference in modelled outputs resulting from the use of various SOC databases. a.) Amalgamated databases; b.) National Soils Institute Inventory database; c.) RSS database; d.) Countryside survey database 1998; e.) Countryside survey 1978 database; f.) Wallington database

When the model incorporating climate warming is run, and the effects of climate warming are considered in addition to land-use change, a similar pattern over time is observed (Figure 6.12), with a median flux into the soil of 64 Tg C (inter-quartile range: source of 48 – sink of 157 Tg C).

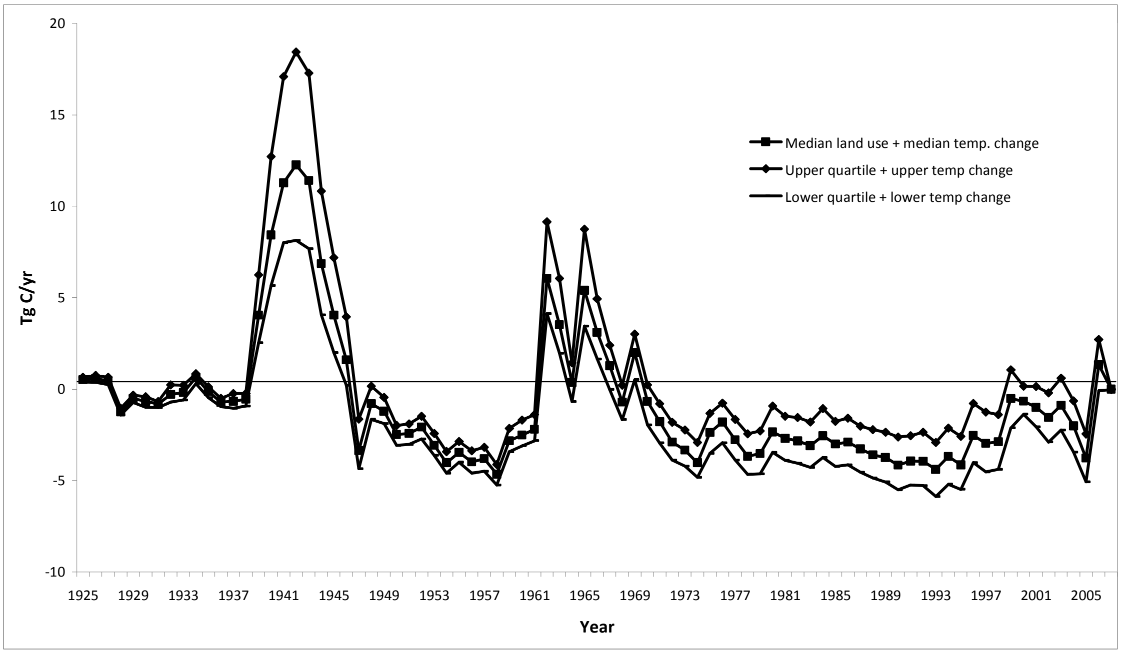

Figure 6.12 The modelled flux of UK SOC from 1925-2007 when both land-use change and climate change are considered. A positive flux represents a flux from the soil to the atmosphere; a negative flux represents a flux from the atmosphere into the soil.

Although this sink is not of the same extent as of that when only land-use change is considered, and although it could in fact be a source, a median yearly sink rate of 0.77 Tg C/yr (interquartile range: source of 0.58 Tg C/yr – sink of 1.89 Tg C/yr) indicates that SOC emissions over the study period are likely to have been much lower than the 12.50 Tg C/yr estimated by Bellamy et al. (2005) for the 1978 – 2003 period, and also lower than the maximum loss due to climate change (2.50 Tg C/yr) estimated by Smith et al. (2007). Our results suggest that the carbon sink provided by land-use change has more than offset the carbon emissions resulting from climate change.

6.4 Discussion

From initial observation of the changing land-use from 1925 - 2007 (Figure 6.6), one might expect that the UK will have been a source of SOC to the atmosphere, due to the decrease in area of permanent pasture, increase in area of urban land, and very slight increase in arable area. The results presented above, however, indicate that this was not the case, and that the UK's soils have been a net sink of carbon. Reference to Figure 6.9 shows that the majority of this sink is the result of land-use change into woodland, and conversion of arable land to permanent grassland. The greatest SOC loss to the atmosphere in 1942 of 12.28 Tg C can be explained by the large increase in arable land as a result of government persuasion to plough up large areas of permanent grassland (Holderness, 1985) during World War II. Two other large fluxes to the atmosphere evident in Figure 6.8 for the years 1961 and 1964 could also correlate with a second phase of arable expansion. Figure 6.9, however, shows that some of these emissions were also the result of urban expansion. The modelling of individual land-use change shows that there were a number of SOC fluxes occurring at this time, and that some of the emissions from urbanisation and arable expansion were being compensated for by the sinks associated with land-use change to woodland and conversion of arable land to grassland. Had the latter changes not occurred, the flux to the atmosphere would have been of an even larger magnitude. The greatest gain in SOC in 1993 could be in part the result of the introduction of set-aside in the UK, in agreement with the situation in the US, where set aside has been responsible for a net increase in SOC (Ogle et al., 2009). Although voluntary set-aside began in 1988, it was in 1993 that the Arable Area Payments Scheme came into force (Adger and Subak, 1996). Investigation into the fluxes caused by arable to grass conversion (Figure 6.9), however, reveals that although there was an amount of sequestration caused by this land-use change, it equated to only 1.38 Tg C. This is similar to the figure of 0.8 Tg C quoted by King et al. (2004) for conversion of arable land to set-aside. The large emission in 2005, and the general decreasing trend in carbon sequestration from 1994 to the present day can be explained by changes out of grassland, out of woodland and land-use changes into urban land (Figure 6.9).

For comparison with the results from previous studies into SOC losses, Bellamy et al. (2005) estimated a loss of 12.5 Tg C/yr due to climate change for the period 1978-2003. Smith et al. (2007) predicted the loss due to climate change to be (at maximum) only 20% of this, therefore ~ 2.5 Tg C/yr. This model predicts a 4.81% net gain in SOC stocks between 1978 and 2003 due to land use change, representing a SOC gain of 96.57 Tg C (3.86 Tg C/yr). The model used in this study predicts a gain of 28.00 Tg C over the period 1984-1990, in comparison to Howard et al.'s (1995) estimated loss of 32.64 Tg C. Therefore, it would appear that land use change over each of these periods is capable of offsetting much of any predicted loss due to climate change.

The results of this modelling study not only provide information on how much of our GHG emissions over the last century can be ascribed to SOC, but may also help guide our future land-use change decisions. These results emphasise the importance of considering simultaneous land-use change decisions, and how some SOC fluxes may be counteracted or added to by other land-use changes then or within a few years. This is important in terms of reacting to the current increases in GHG emissions, and the need to realise that what occurs in a single year can cause large emissions, and that these should be avoided when there is limited time to act to curb GHG losses. The model allows us to assess land-use change contribution to CO_2 emissions or sequestration on a yearly basis from 1925 to 2007, therefore revealing carbon fluxes that would otherwise be obscured if only the change in SOC flux at the beginning and end of the period were measured. The extent of the fluxes caused by some land-management changes (e.g. the emissions of 1942) may have been missed if this approach had been taken, due to counteraction by fluxes in the opposite direction in later years. Although Tomlinson and Milne (2006) assessed soil carbon changes from 1939-2000 in Ireland, they did this assuming that the total changes in land-use over this period occurred at an equal rate over the period. Such an approach does not reveal the consequences of a rapid flux, such as that of 1942 in the UK, and cannot inform our understanding of the short-term effects of any such rapid changes in the future.

These results help us to assess the contribution of SOC to the UK's total GHG emissions. The UK's current industrial emissions are reported to be approximately 150 Tg C/yr (Bellamy et al., 2005; Ciais et al., 2009). Reference to Figure 6.8 shows that the sink observed was not a linear change in carbon stocks over the 82 year time period, as if this was the case it would represent an increase in SOC stocks of 1.92 Tg C/yr. This suggests that on average, over the entire period, SOC has not contributed to UK emissions, and that land-use changes have in fact sequestered carbon from the atmosphere. As discussed earlier however, assessing the contribution over such a large number of years does not reveal the extent of the contribution of some large yearly fluxes. The loss of 12.28 Tg C in 1942 represents more than 8% of the UK's current industrial emissions. This figure is much more relevant to our actions in current times, indicating that such land-use changes now could have severe consequences in the context of growing international pressure to reduce GHG emissions over a short time period. This loss, however, represented only 0.82% of the UK's estimated SOC stock at that time, and is therefore significantly smaller than the 10% change in global SOC stocks that would be needed to represent 30 years of global anthropogenic CO_2 emissions (Kirschbaum, 2000). The sink of 4.31 Tg C in 1993 represents 2.87% of the current UK's industrial emissions. This indicates that although similar land-uses changes in the future could be made to offset some of the industrial emissions, many other changes besides land-use change must also be implemented if the targets of 34% reduced CO_2 emissions by 2020 are to be met.

The results from the additional model run in this study which incorporated climate change can help to establish the contribution of SOC to UK GHG emissions when the likely effects of climate change on SOC are included. When including climate change, we predict a flux of carbon from the atmosphere to SOC. The median annual flux over the period 1925-2007 with climate change included equates to 0.77 Tg C/yr, contrasting with the findings of Bellamy et al (2005). The results of this study suggest that the SOC sink provided during this period by changes in UK land-use, have more than compensated for their quoted losses resulting from either climate change or agricultural land management change. When the effects of climate change are incorporated into the model the flux to the land still remains. The effects of increased productivity under a changing climate, and potential for improved agricultural technology on counteracting a tendency for climate change to speed decomposition, and thereby enhance SOC loss, was also examined by Smith et al. (2005), who showed that increasing SOC sinks are possible under a changing climate.

This research also allows us to assess the extent of the UK's carbon emissions in relation to other countries, and the land-use changes that they have made. The loss of 12.28 Tg C in 1942 is three to four times greater than the loss of SOC predicted from SOC models looking at the conversion of forest land to arable land in Brazil (Maia et al., 2009). It is estimated that throughout the period 1985-2002 3.74 Tg C/yr were lost from the soil in Brazil (Maia et al., 2009), though the largest losses from deforestation are in the lost vegetation. Although the loss estimated in this study for the year 1942 only occurred for one year, and is not a continual situation in the UK, it emphasises the extent of the losses possible from UK land-use change, when the levels emitted from Brazil's rainforest deforestation are of great concern on a global scale.

Caution needs to be taken when assessing the results from all model outputs, as Wutzler and Reichstein (2007) argue that using an observed carbon stock in a soil carbon model to represent equilibrium is incorrect because it is based on the assumption that the soil sample represents a soil at equilibrium. The likelihood that the soils are unrepresentative of soils at equilibrium is however reduced with the scale of the sample size used in this study, and the use of such a large database in this study provides the results with an element of authority and should be considered as a more accurate model than those using much smaller databases on which to base their equilibrium SOC stocks (e.g. Maia et al., 2009). Similarly, non-equilibrium effects are far less important over long time periods, such as the decadal timescales studies here.

The results show that the use of a regional database or a country specific database when attempting to estimate SOC fluxes on a scale as great as the UK will provide inaccurate results (Figure 6.10). When predicting fluxes for the UK the amalgamated database was considered the best source, as it is expected that the number of samples from different soil types will be representative of the area of these soil types in the UK. This is in comparison to a regional database where a large area of highly organic or mineral soil may skew the SOC values. This is represented by use of the Wallington database for predicting the UK SOC flux, where a large area of forestry exists on organic soils (Bell and Worrall, 2009). In the case of the UK however, the approach used in this study is deemed more than satisfactory, and could not be improved given the land-use change information available.

Although the SOC transition times used in this modelling study are only estimates, it is believed that these are the most accurate available, having been estimated using evidence from over 20 previous studies for both SOC losses and gains. In an ideal world transition times would be measured following land-use change, however the slow rate at which SOC adjusts to this change makes long-term trials difficult to conduct, and is the reason why such information is lacking (Ogle et al., 2009).

When interpreting the results it has been assumed throughout that any losses or gains in SOC are the result of either carbon sequestration from the atmosphere or a release of carbon to the atmosphere. One final point of caution is that some of this SOC loss following a land-use transition may not in-fact have been emitted to the atmosphere, and could actually have been lost to surface waters as DOC, dissolved CO_2 or leached into deeper soil layers. Although there is no evidence for increased dissolved CO_2 losses over time from the UK (Worrall et al., 2007) there is extensive evidence that DOC flux from the UK has increased. Worrall et al. (2009) has shown that DOC flux from the UK has increased from 0.8 Tg C/yr in 1975 to a peak of 1.9 Tg C/yr in 2003. An increase in carbon losses via DOC export is of the order of 1 Tg C/yr, or more than the amount suggested by Smith et al. (2007) as the maximum SOC loss that could be attributed to climate change. Although the extent to which this is likely to have occurred is currently unknown, it implies that any of the quoted figures relating to carbon sequestration over this period should be adjusted upwards (to a larger carbon sink), as losses to the atmosphere may not be as great as initially thought.

6.5 Conclusions

Although there are still many uncertainties involved in modelling the impact of land-use on the UK's SOC stock, and SOC fluxes to and from the atmosphere, this study reveals the order of magnitude to which land-use is affecting total atmospheric CO_2 levels, providing an insight into the importance of our future land-use change decisions. The model created in this chapter can be used by the NT as a tool to investigate the potential impacts of any future land-use change on atmospheric carbon levels, and can therefore guide future land-use change decisions in order to meet the aims of the organisation as outlined in Chapter 1.

Chapter 7

Charcoal addition to soils in north east England: a carbon sink with environmental co-benefits

A reformatted version of this chapter has been accepted for publication in the journal *Science of the Total Environment*.. This paper is co-authored by Fred Worrall. I carried out all of the data collection, analytical work and data processing. I wrote the manuscript in its entirety and passed it on to the co-author for feedback. Fred Worrall provided continuous support and guidance throughout, giving critical feedback and help in directing the discussion of the manuscript.

7.1 Introduction

Research into the impacts of land-use on %SOC is largely consistent in its findings regarding arable land, concluding that losses of carbon from the soil will occur on conversion into this land-use, and that gains in soil carbon will be achieved upon conversion out of arable land (Post and Kwon, 2000; Guo and Gifford, 2002). These results are supported by the modelling results in Chapter 6, indicating a large loss of carbon to the atmosphere following conversion from pasture to arable land in 1942. If land-use change is to be made solely for the purpose of increasing carbon sequestration in soils, and reducing carbon emissions, then the results of Chapter 6 would suggest that no land should be converted to arable, and that all current arable land should be converted to an alternative use. In order to achieve the aims set out in Section 1.1.1, of creating new carbon sinks and new carbon stores, the NT could remove all arable land from its estates, however this will not meet the requirements set out in Section 1.1.2, as it must be recognised that arable farming is a way of life for many farmers, and that the growth of arable food products will provide economic returns for many. It was therefore considered a priority of this research to identify ways in which arable farming could continue to be practised, whilst limiting the losses of carbon to the atmosphere, and sequestering carbon into soil. Arable fertilisation techniques, crop rotation schemes and tillage methods are thought to be variables which can be altered to reduce carbon losses from soil under arable land, however the results of such trials are variable (Chapter 3) and the carbon sink benefits may be small in magnitude (Hutchinson et al., 2007). The results and research presented in Chapters 2 and 3 suggest that arable SOC stocks can be altered by land-management change, however there are still elements of uncertainty which must be overcome before any firm land-management change can be made. For arable farming to continue, other methods of land-management change within arable land-use are sought whilst these uncertainties are resolved.

As indicated in Section 1.4 there is an emerging area of research associated with the incorporation of biochar (charcoal produced as a by-product of energy generation during the pyrolysis of biomass (Tenenbaum 2009)) into soils to sequester carbon, with the potential to also improve soil quality, increase crop productivity, and hence sequester atmospheric carbon in the process. If biochar can live up to these potentialities it could be used as a tool to counteract the GHG emissions that the NT are attempting to offset (Chapter 1), whilst allowing arable land-use to continue. This is a very attractive option for the NT and other land-owners, but before such a technique can be implemented, research into its effects not only on carbon sequestration, but also on water quality, crop productivity and fertiliser leaching is required, to meet the aims outlined in Section 1.1.2.

The prospect of using biochar as a tool to mitigate climate change, whilst simultaneously allowing the continuation of arable farming (and relieving potential food shortages) on NT estates is highly attractive. Many attempts to mitigate against these potential problems are underway, however in the majority of cases each issue is considered as a separate entity, and the mitigation methods are implemented solely to solve the issue of concern (e.g. carbon capture and storage in the oceans to sink carbon (Ametistova et al. 2002; Praetorious and Schumacher 2009) or genetically modified crops to increase food production (Thomson 2003). In an ideal world however, any mitigation approach would tackle such issues together and offer co-benefits not only to offset costs and save time, but also to reduce or offset risks associated with such methods.

A decision was made that research into biochar application to soils should be expanded to include the impact of incorporation into more organic-rich soils. If the carbon sequestration potentialities relating to biochar incorporation into arable land also apply when incorporated in organic rich soils (located under forestry plantation on the Wallington Estate), the area of land to which biochar can be utilised will greatly increase (Figure 1.2). This research, along with that of incorporation into soils under arable land is the focus of this chapter.

7.1.1 Biochar: a review of the literature

The burial of biochar in soil is, according to many researchers, a potential way to sink carbon and lock it away from the atmosphere, whilst simultaneously acting as a fertiliser and increasing PP (Spokas et al. 2009; Major et al. 2009; Gaunt and Lehman 2008). The burial of biochar in soil in other studies has also been shown to stabilise (Steiner et al. 2008), or even increase SOM (Kuzyakov et al., 2009; Cheng et al., 2008), therefore potentially improving soil quality and reducing soil degradation. The high porosity of biochar also suggests that its application could improve soil water holding capacity (DeLuca et al. 2009), and therefore mitigate against potential drought conditions in certain areas of the world in a predicted warmer climate. The potential fertilising effect of biochar (due to nutrient and water retention and liming) could reduce the need for artificial fertilisers, not only lowering costs to the farmer, but also bringing further benefits in the form of reduced CO_2 emissions from the reduction in artificial fertiliser production (Gaunt and Lehman 2008). Other potential benefits include: reduced water pollution by fertiliser leaching; a reduction in heavy metal leaching (Cao et al. 2009); and financial savings from an increase in soil pH and a resultant need for less lime.

In contrast to the attractive prospects of biochar as outlined above, there are several streams of conflicting research in the literature, suggesting a need for caution. Some of these are highlighted in a recent article by Wardle et al. (2008), which suggests that the benefits may not be as great as initially thought. Trials have shown that the addition of biochar may enhance microbial activity and cause a greater loss of the carbon already present in native SOM. Similar results have also been found by Hamer et al. (2004), implying a loss of CO_2 to the atmosphere as a result of biochar addition- the opposite of what is trying to be achieved. The suggestion of increased microbial activity is highlighted in several other studies that claim microorganism activity is encouraged by biochar addition (Mathews 2008; Lehman and Rondon 2006). There are however other reports suggesting that an increase in microorganism activity could in-fact increase SOM as a result of increased humus formation (Gundale and DeLuca 2007). Other evidence for enhanced OM breakdown with biochar addition comes from Rogovska et al. (2008), who found increased CO_2 emissions, but no loss of the added biochar. Oxidation of the biochar itself is debatable, and there are many questions relating to its stability (Lehman et al., 2006; Nguyen et al., 2009; Knicker 2007), with reports of both rapid (a half-life of less than 100 years) (Bird et al. 1999) and slow decomposition (Shindo 1991).

In terms of the effect of biochar addition on PP, the results are again uncertain. Crop yields have increased with biochar application rates of up to 140 000 Kg/ha, however it is not known if any negative affects on PP will result if application rates exceed such high doses (Lehman et al., 2006). Studies have also found contrasting results depending on whether fertiliser treatments have also been applied (Asai et al., 2009).

There is also a noted lack in the literature of research into biochar's effects on nutrient retention (Lehmann et al., 2003). The Terra Preta soils of the Brazilian Amazon stimulated the initial interest in biochar application to soils as they are extremely fertile (Steiner et al., 2008) and contain large amounts of SOC. Their extreme fertility suggests that the charcoal may be retaining nutrients and acting as a natural fertiliser. These Terra Preta soils were formed when agricultural wastes were smouldered and incorporated into the soil by earthworms during pre-Columbian Amazonian times. Although possible, and likely in terms of positively charged nutrients such as potassium, calcium and magnesium, the suggestion of nitrate retention by negatively charged charcoal is very unlikely due to the negative charge of nitrate and the fact that it is easily lost from negatively charged soils (Santibanez et al., 2007). Despite this there are several studies which argue that nitrate retention increases with increasing biochar addition to soil (Akinor et al., 2001), however the mechanisms responsible for this are unclear.

The conflicting results and level of uncertainty currently given to biochar application in part stems from the different environments under which trials have been conducted, and comparison of studies carried out in very different environments and on different soil types. The majority of research has been carried out on agricultural soils low in carbon content and in tropical climates. Assessment of the effect of adding charcoal to more organic rich soils in temperate climates is therefore also needed, as it is possible that these could behave in a very different manner.

7.1.2 Chapter aims

Although the potentialities of biochar described in Section 7.1.1 and Section 7.1.2 present its use as a positive and promising opportunity, it must be realised that each potentiality needs to be confirmed or refuted before biochar application can proceed on a large scale, to different soils and crops in varying environments. The aim of this chapter was to assess the implications of applying lump-wood charcoal (used in this study as a substitute for biochar) to low carbon soils in arable land-use, and more organic rich soils under plantation forestry, typical of soils and land-uses on the NT Wallington Estate (Section 1.1.1 and 1.1.2).

This chapter assesses the impact of biochar on several aspects of the carbon balance, as well as the impact on fertiliser use and the environment. This was in line with the aims of the NT set out in Section 1.1.2 and Section 1.3. These aims suggested that the uptake of biochar application should not be judged solely on the permanence and stability of biochar (e.g. Fowles 2007), and the size of the carbon sink, or solely on its use as a fertiliser (e.g. Yeboah et al., 2009). This study takes the approach that even if there is a loss of SOM as a result of charcoal addition, or a loss of the added charcoal itself, these losses should not be used as an argument against biochar addition unless the net carbon balance is positive (a net release to the atmosphere), or unless water quality and crop productivity are negatively impacted. A loss of SOM, if found, may only be one negative effect amongst a number of positives, resulting in charcoal application still acting as a carbon sink and still providing crop productivity benefits. The aim of the study was to weigh up any carbon sinks/emissions against environmental costs/gains associated with nutrient leaching, and economic losses/gains related to fertiliser/lime use etc. To achieve the objectives of this chapter it was aimed to measure the following:

- Losses of carbon to the atmosphere as NER
- Carbon sequestration in the form of PP
- Losses of carbon in leachate as DOC
- Leachate properties: anion concentrations, pH, electrical conductivity

7.2 Materials and Methods

7.2.1 Study site and location

The study uses lysimeters as these allowed for replication within a factorial design. The large majority of lysimeter and incubation studies have been carried out in the laboratory where conditions have been controlled and temperatures and water levels kept constant (Liang et al., 2008; Kuzyakov et al., 2009). This study differs in this respect, as the free-standing lysimeters were placed outdoors, where they were subject to the natural, prevailing, external temperature and rainfall conditions, in the same area of the country that biochar will be incorporated if application to land proceeds. This was deemed necessary as there is a need to evaluate how biochar amended soil will respond to real variations in environmental conditions.

The study was undertaken at Durham University (55°47'N, 01°34W), approximately 64 km south of the source of the study soils on the NT Wallington Estate (Bell and Worrall 2009). Conditions at both the soil source and study locations are very similar, typical of the UK's temperate climate. Soil was collected from the top 20cm of an arable field on the Wallington estate and mixed to form a bulk sample. The specific field was chosen as it had been in its current arable land-use for at least the last 28 years, meaning that its SOC content should be in equilibrium and should not be adjusting to land-use change which could obscure the outcomes of this study. The soil series from which the soil was taken (Brickfield - Hollis 1975: Jarvis 2002; Payton and Palmer 1989 and Table 7.1 for soil descriptions) had the greatest aerial coverage under arable land-use on the Estate, and was chosen to represent how soils typical of arable land-use and with low %SOCs would respond. The same criteria were chosen when locating an area for collection of the organic rich forest soils (Wilcocks soil series- Hollis 1975: Jarvis 2002; Payton and Palmer 1989 and Table 7.1 for soil descriptions).

Soil series	Referred to in study as:	Typical %SOC (0-20cm)	Typical bulk density (g/cm^3) (0-20cm)	Typical profile description (0-20 cm)	% cover on Estate
Brickfield	Arable soil	4.19	1.3	Sandy clay loam. Very dark greyish brown with ocherous mottles	16.13
Wilcocks	Forest soil	7.49	1.03	Black humified peat/organic sandy clay loam	24.30

Table 7.1 Study soil descriptions and general properties taken from soil survey memoirs.

The mass of soil to which charcoal was added was chosen based on the typical bulk density of Brickfield and Wilcocks soils respectively. The soil from the arable field had previously been fertilised with Ammonium-Nitrate fertiliser by the farmer. All roots and crop residue remaining in the soil following collection from the field was removed by sieving through a 2mm sieve. This gave the total bulk sample a uniform content, and resulted in controlled experimental conditions. The soils sampled were homogenised in order to remove any potential confounding effects of local scale variations in SOM. This study does not attempt to make an assessment of any spatial distribution of charcoal application effects on the soil properties in the field sites chosen.

7.2.2 Experimental design

Although the initial aim of this research was to assess the impacts and possibilities of adding biochar produced from short-rotation willow coppice or farm waste to the differing soil types studied, the lack of availability of these resources meant that equivalent quantities of lump-wood charcoal were used as a substitute. The charcoal was manually crushed to a powder using a pestle and mortar – as per previous studies (Steiner et al. 2008). The large majority of the powder was < 2mm, however some pieces remained slightly larger due to time constraints: Lehman et al. (2003) found no significant difference in the results from trials looking at crop productivity when ground charcoal was used compared to charcoal pieces. The quantities of charcoal applied in this study were chosen on the basis that the NT's aim was to counteract the net annual emissions from their Wallington Estate. Total annual net carbon emissions from the Estate were calculated at 794.54 tonnes C/yr (based on soil carbon flux, biomass carbon flux, visitor travel and energy use), therefore if the aim is to be carbon neutral they will need to sink 794.54 tonnes C/year. This figure was then divided by the area of arable soils on the estate to which biochar could be applied, giving an application rate of 2650 kg C/ha/yr. Due to the use of lumpwood charcoal as a substitute in this study, this value was then adjusted to allow for an estimated lumpwood charcoal carbon content of approximately 80% (Lin and Hwang 2009), giving an application rate of 3312 kg/ha. An application rate of 6250 kg/ha was then chosen to approximate the lower level values used in previous studies, and to therefore make the Estate carbon neutral for approximately 2 years. It was assumed that the quantity applied would not undergo any decomposition and lead to any native SOC loss, nor would it lead to increased PP and carbon sequestration. This assumption was made as the effects of addition were not yet known, and biochar application rates required to make the Estate carbon neutral could be adjusted accordingly once the results on charcoal decomposition, SOC loss and PP were available at the end of the trial. Two further treatment levels were chosen to assess the potential benefits or negative impacts of applying this rate continuously on an annual basis, or alternatively as a one-off application making the Estate carbon neutral for many years. The low rate of charcoal added as treatment level 1 also allowed investigation of the extent of any crop productivity benefits at such a small dose of charcoal in the early years of addition. This would reveal whether any crop productivity benefits would be sufficient to encourage the uptake of such a technique in these early years of research. The two higher levels, as well as allowing assessment of continual application at Wallington, were also chosen as PP in previous research has increased at slightly lower levels than these, however possible negative affects at these higher levels has not been assessed. Lysimeters containing 0 kg/ha charcoal were chosen to act as a control (Table 7.2).

Charcoal treatment	No. of years equivalent application	Kg charcoal/ha	Kg C/ha	Kg biochar/ha
0	0	0	0	0
1	1.8	6250	5000	10 000
2	18	62 500	50 000	100 000
3	26	87 500	70 000	140 000

Table 7.2 Charcoal treatment levels

Lysimeters were constructed following the design of Worrall et al. (1999), with each consisting of a 35cm length drainage pipe (diameter 15cm), with a wire/nylon mesh bottom. This enabled water to filter through whilst at the same time supporting the large volume of soil and limit the removal of any soil particles. Each pipe contained unmodified soil (containing no charcoal) in the bottom 13 cm followed by the soil/charcoal mixtures in the next 20cm, leaving a 2cm gap at the top. Charcoal was only added to the top 20cm of soil as it was assumed that this is the plough depth to which any biochar would be added, and therefore is representative of field conditions. The pipe was placed onto a collar into which a glass funnel was inserted, allowing water to drain from the soil. The water entered a 2 litre plastic bottle contained in the bottom half of the lysimeter.

A completely randomized four factor experiment began in October 2008 and ran for a total of 26 weeks until May 2009. Time limitations meant that the trial did not include the summer months. The four factors were soil type (with 2 levels - arable, forestry);

vegetation (with 2 levels – planted with perennial ryegrass-seed; or no ryegrass-seed); charcoal treatment level (Table 7.2) and time. The factor vegetation was however only applied to the arable soils, as the organic rich forest soils on the estate are used to grow coniferous trees and not crops. It was therefore felt that establishing the impact on grass growth for this soil type would not be indicative of the impact on plantation forestry trees, and further research into this is required.

Each soil/charcoal or soil/charcoal/vegetation combination was duplicated, resulting in a total of 24 lysimeters. Grass seed was sown into the lysimeters containing arable soil on 22nd October 2008. 0.8835 g of grass seed was sown into each lysimeter based on planting instructions of 50 g per square meter.

7.2.3 CO_2 monitoring

NER was monitored weekly from the soil surface using a dynamic dark closed chamber and Infra-red gas analyser following the method described in Section 4.2.4.1. This allowed the change in CO_2 concentration within the chamber to be measured, and hence the CO_2 efflux from the soil surface. Analysis took place between the hours of 1400 and 1600 on the day of initial measurement and subsequently every week following. This was in an attempt to minimise any effects associated with daily temperature, moisture and light variation which could obscure the results due to 24 hour variability in soil carbon flux (Mielnick and Dugas 2000). Measurements were taken weekly for the entire trial period of 26 Weeks. For the lysimeters planted with grass-seed PP was monitored from week 13 to week 26 by monitoring NEE as well as NER and subtracting the value for NEE from the NER value (as described in Section 4.2.4.2 and Section 4.2.4.3).

7.2.4 Leachate monitoring

Water was collected at regular frequencies depending on climate and storm events, and the day of collection was recorded. Due to a variation in climate throughout the study period this meant that the sampling time varied from 1 week intervals (Following heavy rain and snowfall) to 4 week intervals (following warm, dry conditions). This resulted in a total sample size of 16 water samples from each of the 24 lysimeters. Samples were filtered through 0.45 µm filters. Water volume, pH and conductivity were analysed immediately in the lab and subsamples were taken and frozen for later nutrient and carbon analysis. Conductivity and pH of the leachate were measured using electrodes, and DOC analysis of all samples was performed using the colorimetric method of Bartlett and Ross (1988) described in Section 4.2.6. Analysis of nitrate, sulphate, chloride, bromide and fluoride was done by ion chromatography following the method described in Section 4.2.6.

After a total period of 222 days the trial was ended and all above and below-ground biomass was harvested from the vegetated lysimeters. Following the method of Lehman et al (2003) below-ground root biomass was removed by hand and dried overnight in an oven at 70 °C.

7.2.5 Statistical Analysis

The sampling design conducted in this study could be considered as a 4 factor experiment. The four factors were (i) soil type, (ii) charcoal treatment level, (iii) vegetation and (iv) time. No covariates were included in the analysis of the role of charcoal treatment level on NER, however when assessing the affects of charcoal treatment level on leachate properties the following were considered as covariates: leachate volume, pH, electrical conductivity, DOC content and nitrate content. The factors were entered into an ANOVA GLM as categorical variables, and the covariates as continuous variables using Minitab statistical software. As each soil type and vegetation type were analysed separately it meant that only two factors: charcoal treatment level and time were entered into the GLM for each analysis. NER data was analysed by ANOVA and the leachate property data by ANCOVA. Results were considered statistically significant if $p \leq 0.05$ (95% confidence interval), and the results of ANOVA and ANCOVA were post-hoc tested using the Tukey test. The proportion of the original variance explained by factor and covariate was then calculated using the method of Howell (2002).

To assess the impact of charcoal application on the total annual carbon balance a mean NER value and DOC leachate volume was calculated for each charcoal treatment level and these were then converted into emissions of kg C/ha/day and the charcoal additions and PP values as sinks of kg C/ha/day. This assumed that the mean results from the 26 week trial period were representative of annual results. It is recognised however that this is a limitation which could result in an annual under estimate of CO_2 respiration and over estimate of DOC flux, due to the trial missing out the warmer, drier months of the British climate, when respiration can be expected to be higher and leachate volume lower. Despite this limitation it was believed to be the best way to compare the effects of charcoal treatment, as it is believed that the relative effects observed during the trail period will not differ with season.

To assess the impacts of charcoal application the three soil/vegetation combinations were assessed separately. The effect on NER, PP, DOC leaching, nitrate leaching, leachate pH, leachate electrical conductivity and leachate volume were assessed separately and discussed in combination in the discussion section.

7.3 Results

The direction of change in each flux or parameter in relation to the control (when 0 charcoal is applied) with each treatment and on each soil/soil-vegetation type is shown in Table 7.3.

All NER measurements made in this trial can be found in Appendix 7 under the heading 'CO_2 flux data'. All measurements relating to water chemistry and leachate properties are located in Appendix 7 under the heading 'water chemistry'.

	NER			DOC			PP			Biomass			Nitrate leaching			pH			Water holding capacity		
Charcoal Treatment	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Bare arable soil	↑	↑↑	↑ C; S	↑	↑ D	↑↑ S							↓↓	↓↓	↓↓ N; S	↑	↑↑	↑↑ PH; S	×	×	× S
Vegetated arable soil	↓	↓ C	↑ S	↑	↑ ↑	↑ D; S	↑	↑	↑ P; S	↑	↑	↑ P; S	↓	↓	↓↓ N; S	↓	↑↑	↑ PH; S	×	×	× S
Forest soil	↑↑	↑	↑ C; S	↑	↑	↑ D; S							↓	↓	↓ N; S	↑	↑↑	↑↑ PH; S	×	×	× S

Table 7.3 Changes in NER, DOC, PP, total biomass, Nitrate leaching, pH and water holding capacity on different soil/vegetation combinations with charcoal application. ↑: Increase in magnitude relative to zero char treatment; ↑↑: Statistically significant increase in magnitude relative to zero char treatment; ↓: Decrease in magnitude relative to zero char treatment; ↓↓: Statistically significant decrease in magnitude relative to zero char treatment; C: Preferred application for CO_2 flux benefits; D: Preferred application for DOC loss benefits; P: Preferred application for PP benefits; N: Preferred application for Nitrate leaching benefits; PH: Preferred application for pH benefits; S: Preferred application for C sink benefits; 1-3:Charcoal treatment level 1, 2 or 3; ☐:Preferred application for both the parameter in question and C sink

7.3.1 NER

On the un-vegetated arable plots NER increased with increasing charcoal treatment from level 0 to level 1, and again to level 2; however there was then a decline to level 3. There was a statistically significant difference between charcoal treatment level 0 and 2, but the change from level 0 to level 1 was not statistically significant, nor was the decrease from level 2 to level 3. There was a statistically significant interaction between charcoal treatment level and time, indicating that in some weeks charcoal treatment had a significant affect on NER, whereas in other weeks it did not. This can be seen in Figure 7.1, where the significant difference appears only to have occurred in week 1.

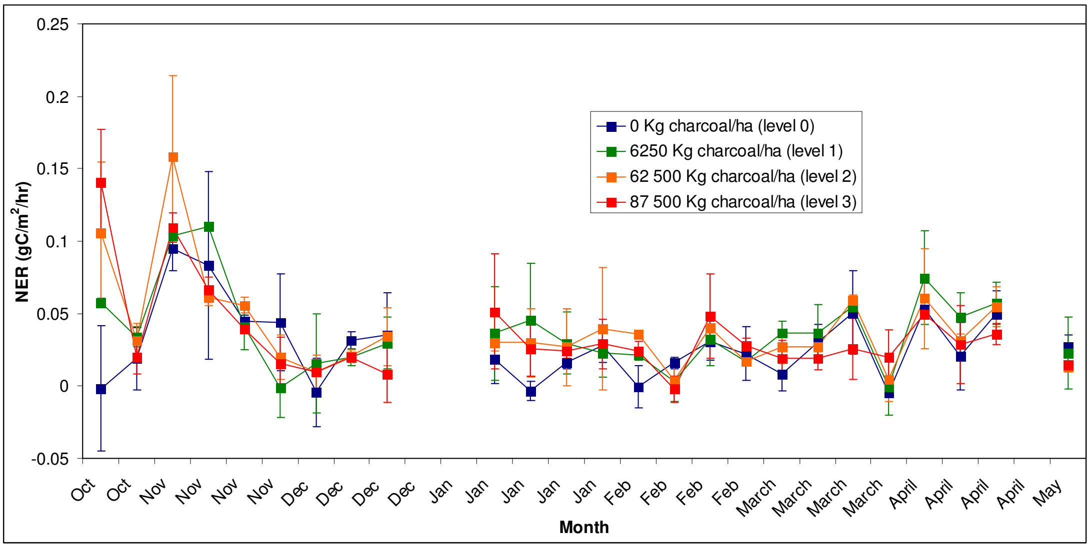

Figure 7.1 The effect of charcoal treatment level and measurement week on NER with charcoal application to bare arable soils. Symbols represent the data means and vertical bars represent standard deviations.

When both charcoal treatment and time, and the charcoal treatment/time interaction were known, 79.80% of the variation in NER could be explained; however the proportion of original variance explained by time (65.00%) was much greater than that of charcoal treatment (2.55%), or charcoal treatment/time interaction (7.69%). This indicates that temperature, precipitation and other variables changing on a weekly basis are much greater controls on NER, and that the effect of charcoal treatment is very small in comparison. Comparison in terms of a daily CO_2 flux from the soil is shown in Table 7.4.

Soil/soil-vegetation	Treatment	Charcoal C addition (kg/ha/day)	PP C addition (kg/ha/day)	CO_2 loss (kg/ha/day)	DOC loss (kg/ha/day)	Avoided loss (kg/ha/ day)	Net C sink (kg/ha/ day)	Annual net C sink (kg/ha)	Nitrate loss (total mg during trial)
Bare arable	0	0		6.81	0.42		-7.23	-2638	304.2
Bare arable	1	13.7		9.11	0.62	+7.23	+11.19	+4084	182.8
Bare arable	2	137		9.61	0.48	+7.23	+134.13	+48 957	185.5
Bare arable	3	192		8.38	0.72	+7.23	+189.91	+69 317	112.1
V arable	0	0	3.94	11.02	0.51		-7.59	-2770	300.9
V arable	1	13.7	3.77	9.19	0.57	+7.59	+15.3	+5584	282.7
V arable	2	137	5.58	10.32	0.81	+7.59	+139.04	+50 750	228.3
V arable	3	192	3.69	11.10	0.59	+7.59	+191.59	+69 930	162.7
Forest	0	0		11.81	1.55		-13.35	-4872	5.6
Forest	1	13.7		16.90	1.41	+13.35	+8.74	+3190	3.8
Forest	2	137		13.92	1.45	+13.35	+134.98	+49 267	2.5
Forest	3	192		14.41	1.64	+13.35	+189.08	+69 014	4.5

Table 7.4 Observed carbon losses and gains associated with charcoal addition, and the impact on the net carbon sink and nitrate leaching. Where: –ve is a loss to atmosphere; +ve is a gain to the land; C: carbon; V arable: vegetated arable

The vegetated arable plots showed no statistically significant difference in NER between the 4 charcoal treatment levels. There was however a statistically significant difference in NER caused by time, supporting the results from the un-vegetated plots that variables changing on a weekly basis are much greater controls on NER.

There was a statistically significant increase in NER from charcoal treatment level 0 to level 1 with charcoal application to forest soil, but there were no significant differences between other levels. As with the un-vegetated arable plots there was a statistically significant interaction between charcoal treatment level and time, showing that in some weeks the effect of charcoal treatment was different to other weeks (Figure 7.2).

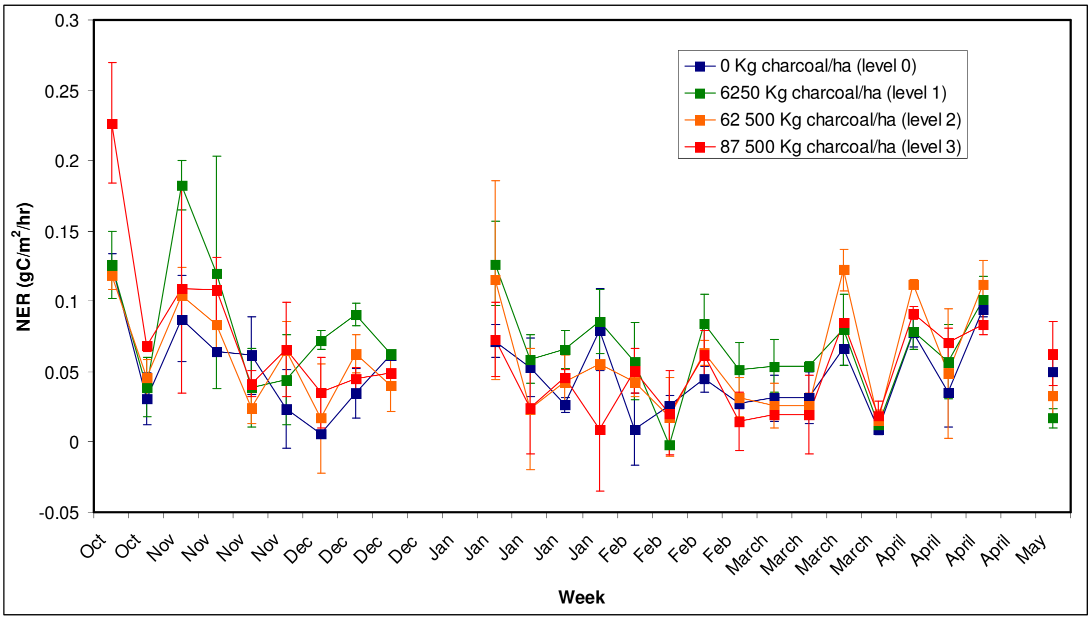

Figure 7.2 The affect of charcoal treatment level and measurement week on NER with charcoal application to forest soils. Symbols represent the data means and vertical bars represent standard deviations.

When both charcoal treatment and time, and the charcoal treatment/time interaction were known, 83.21% of the variation in NER could be explained; however the proportion of original variance explained by time (68.98%) was much greater than that of charcoal treatment (2.98%), or the charcoal treatment/time interaction (7.90%). This supports the results from the arable un-vegetated plots and further indicates that temperature, precipitation and variables changing on a weekly basis are much greater controls on NER than charcoal treatment level.

7.3.2 DOC

The un-vegetated arable plots to which 87500 kg charcoal/ha was applied had a significantly higher DOC leachate concentration than that leached from soils containing 0 kg charcoal/ha (Table 7.3). Week of measurement could however explain more of the variation in DOC leachate concentration than charcoal incorporation. There was a significantly higher DOC leachate concentration from the 62500 kg/ha treatment than from the 0 kg/ha treatment on the vegetated arable plots (Table 7.3), and the effect of charcoal incorporation on DOC concentration did not change with time of year for either the un-vegetated or vegetated arable soils. No other significant differences in leachate DOC concentration were observed between any other treatments for these soils. Although the covariates pH, leachate volume and nitrate concentration could not explain any of the variation in DOC concentration leached from vegetated arable soils, the inclusion of leachate electrical conductivity in the analysis caused charcoal incorporation to become an insignificant explanatory variable, and conductivity and time became the main explanatory factors- suggesting that charcoal treatment level and electrical conductivity are colinear. There were no statistically significant differences in DOC concentration between any charcoal treatments when charcoal was applied to forest soil (Table 7.3). The calculated daily DOC flux from the different treatments on the different soil and soil/vegetation combinations is shown in Table 7.4.

7.3.3 PP

The results of the PP trial refer only to the vegetated arable plots.

In relation to the CO_2 flux measurements made on 11 weekly periods there was an increase in PP between treatment level 0 and all other treatment levels, however none of these were statistically significant. In relation to biomass measurements there were again no statistically significant differences due to charcoal addition.

7.3.4 Nitrate leaching

On the un vegetated arable plots there was a significantly higher total nitrate flux (mg – Table 3 and 4) from the soils containing no charcoal than from all other treatment levels, with a reduction in total mean nitrate flux from 18.7 mg to 12.0 mg (Figure 7.3), and a reduction in mean nitrate concentration from 85.9 mg/l to 50.3 mg/l.

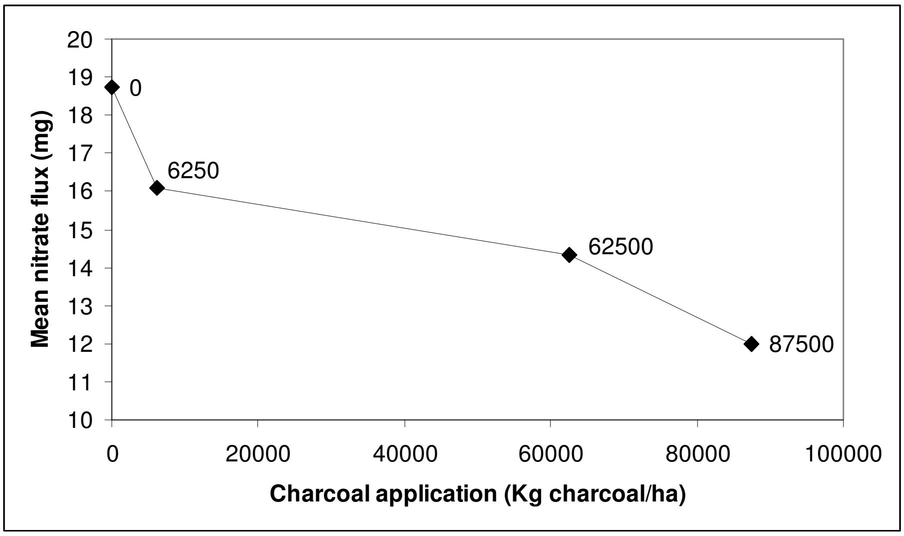

Figure 7.3 A reduction in the mean weekly nitrate content (mg) of leachate with increasing charcoal addition to bare arable soils.

Although there was a lower nitrate flux from the soils containing 62500 kg charcoal/ha and 87500 kg charcoal/ha than from those containing 6250 kg charcoal/ha, neither of these was statistically significant. When volume, conductivity and DOC (mg) were included in the analysis they were all found to have a statistically significant impact on the quantity of nitrate leached. When these covariates were included along with charcoal treatment and time, 85.46% of the variation in nitrate (mg) leaching could be explained. The proportion of original variance explained by volume (l) of 55.74% compared to that of 6.36% for charcoal treatment however shows that the influence of water throughput is much greater than that of charcoal treatment.

Under the vegetated arable plots there was a significantly lower nitrate flux leached from soils where 87500 kg charcoal/ha had been added than from those with no charcoal, and also from those to which 87500 kg charcoal/ha had been added than from those containing 6250 kg charcoal/ha. Quantities of the total amounts of nitrate leached are shown in Table 7.4. Charcoal treatment level and time could together explain 74.37% of the variation in nitrate leaching, however when the covariates volume (l), conductivity and DOC (mg) were included in the analysis they were all found to be significant controls on nitrate leaching, and both charcoal treatment level and time became insignificant.

There were no statistically significant differences in nitrate content leaching between charcoal treatment levels for the forest soils.

In none of the soil types was there a statistically significant interaction between charcoal treatment and time, indicating that the impact of charcoal treatment did not differ from week to week, and that the significant impact on reduced nitrate leaching did not decline over time.

7.3.5 pH

There was a statistically significant increase in pH between charcoal treatment level 0 and level 2, and level 0 and level 3, as well as from level 1 to level 3 on the un-vegetated arable soils. The increase from level 0 to level 3 consisted of an increase in pH from 6.98 to 7.22. None of the covariates other than time were found to have a statistically significant affect on pH. When both charcoal treatment level and time were used as predictors 76.89% of the variation in pH could be correctly predicted.

Under the vegetated arable soils; charcoal treatment level, time and leachate volume were all statistically significant contributors to leachate pH. There was a statistically significant increase from treatment level 0 to level 2 and level 1 to level 2 respectively. The increase from level 0 to level 2 consisted of an increase in pH from 7.09 to 7.21, but there was then a slight decrease to level 3 to a value of 7.18. 81.14% of the variation in leachate pH could be explained using charcoal treatment, leachate volume and time as predictors. The size of effect statistics however show that when other factors are constant charcoal treatment level can only explain 5.92% of the variation, volume can only explain 1.21% of the variation and time can explain 51.47% of the variation. 41.40% of the variation is unexplained.

Forest soils showed a statistically significant increase in pH between treatment level 0 and level 2 and 0 and level 3. The increase from level 0 to level 3 consisted of an increase from a pH of 6.62 to 6.83. As with vegetated arable soils, leachate volume, time and charcoal treatment were all significant controls on pH, correctly predicting 82.21% of the variation in leachate pH. Again the role of charcoal treatment was small when other factors were constant, with the proportion of original variance explained being 7.26% compared to 52.47% for time.

7.3.6 Leachate volume

In relation to all soil types, although there was a decrease in leachate volume with all treatments over treatment 0, none of these were statistically significant. The large variation in leachate volume over the 14 weekly measurements was the result of time, and inclusion of both time and charcoal treatment could explain 80.32% of the variation, although charcoal treatment did not have a statistically significant effect.

7.3.7 Net Carbon Sink

Table 7.4 provides a summary of the net carbon sink benefits attained with each of the charcoal treatment levels when both the losses from NER and DOC are accounted for, as well as the avoided loss which would occur when compared with zero carbon addition.

7.4 Discussion

The reported success of biochar application will depend on the reasons behind application and the degree to which negative environmental or economic effects can be tolerated in light of the potential major gains in terms of a carbon sink.

Although in 8 out of a possible 9 situations there was an increase in NER from the soil containing charcoal compared to that with no charcoal, in only 2 of these situations was the increase statistically significant (Table 7.3). This increase in NER could be the result of an increase in microbial biomass due to the suitable habitat provided by the charcoal (Lehman and Rondon 2006), and therefore increased microbial respiration and OM breakdown. It must be noted however, that only in week 1 out of a total of 25 weeks was the difference in NER statistically significant on the un-vegetated soils, (Figure 7.1) suggesting that any increases in microbial respiration and OM breakdown were a result of disturbance, and that increases in NER resulting from biochar burial in soil will be short lived. Comparison of Figure 7.4 and Figure 7.5 show how the statistically significant charcoal treatment level affect on NER apparent in week 1 is no-longer apparent in week 25.

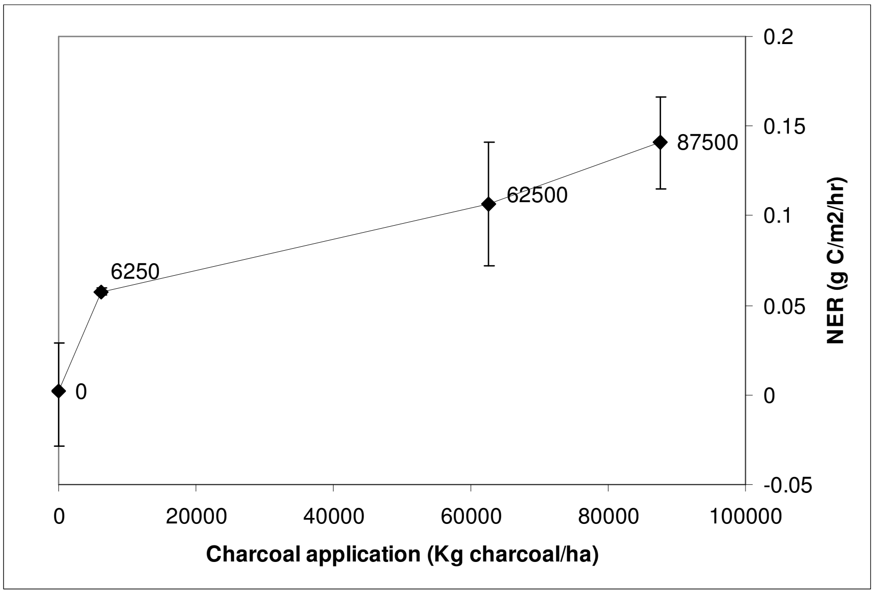

Figure 7.4 Week 1: a statistically significant effect of charcoal treatment on NER. Data points represent the mean values, and the error bars the inter-quartile range. Data point labels refer to charcoal application amounts (Kg charcoal/ha)

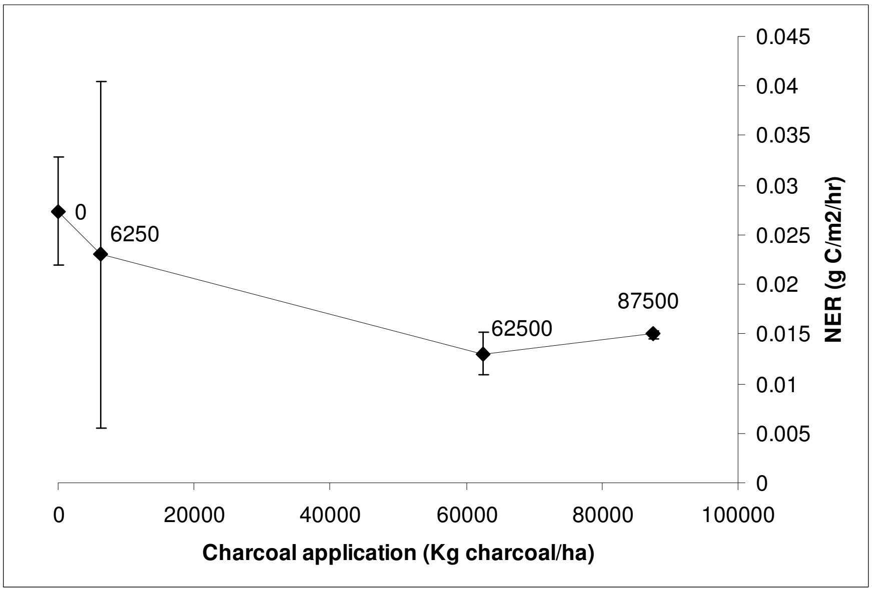

Figure 7.5 Week 25: no statistically significant differences between charcoal treatment levels. Data points represent the mean values, and the error bars the inter-quartile range. Data point labels refer to charcoal application amounts (Kg charcoal/ha)

Comparison of boxplots for all other weeks with week one revealed the same conclusion. Although the other statistically significant difference was found to occur between charcoal treatment level 0 and level 1 on the forest soils, this difference is a result of the total NER flux over the measurement period, and there were no significant differences between fluxes due to charcoal treatment level on a week by week basis (Figure 7.2), further strengthening the suggestion that any effect of biochar on NER is small. Comparison of the magnitude of effect of time compared to charcoal treatment level also reveals that even on the days when charcoal treatment does have a negative effect on carbon emissions, the size of the effect is small in comparison to natural variation in NER occurring on a weekly basis, indicating that any resultant increased carbon emissions are only of a small scale.

The increase in DOC concentration from soil with increasing charcoal content supports this idea of increased microbial biomass and OM breakdown, however it is tiny in comparison to the amounts of carbon being stored by charcoal addition (Table 7.4). Although the DOC increase with increasing charcoal application is very small when it is looked at in terms of carbon loss to the atmosphere, other negative impacts must be considered. Metals and pollutants are said to be mobilised by an increase in DOC, stream life could be reduced due to the decreased in-stream light penetration and water treatment costs could rise (Evans et al., 2005). Although these are very important considerations, the small magnitude of effect of charcoal treatment level indicates that if increased DOC fluxes do occur, these are likely to be obscured by weekly variation and their impact is not likely to be major.

The decrease in nitrate leaching observed in this study could be the result of either microbial uptake of nitrate (immobilisation), or a decrease in nitrification due to the adsorption of ammonium (NH_4^+) to the char, and therefore the production of less nitrate in the first instance. If the hypothesis of nitrate immobilisation is true then this could be a major determinant in the decision of whether to go ahead with charcoal application to agricultural land, if it could result in less nitrate availability for plant uptake. Nitrate immobilisation is supported by the observed increase in CO_2 emissions, as it will result in greater microbial activity; however it suggests that the microbes must be utilising the added C, but this is very unlikely due to the reported recalcitrant nature of charcoal (Kuzyakov et al., 2008; Laing et al., 2008). Although it is often argued that biochar must breakdown eventually, the short period of this trial is unlikely to have allowed carbon breakdown by microbial degradation, and it is unlikely that microorganisms will have been able to utilise this carbon source. This therefore makes the suggestion of nitrate immobilisation unlikely. This is supported by Rogovska et al's (2008) findings of 98 – 109%

black carbon recovery after a 1 year trial. As discussed by Kuzyakov et al. (2008), it is very difficult to determine the charcoal contribution to total CO_2 efflux and further work should focus on assessment of black carbon breakdown in soil, to try and better establish the fate of nitrate. On the other hand, if the decrease in nitrate leaching is the result of NH_4^+ absorption to the charcoal particles, and a resultant decrease in NO_3^- production, then a decrease in CO_2 production may have been expected due to the decreased activity of nitrifying bacteria. Although there was no apparent decrease in CO_2 respiration (there was actually a general insignificant increase), it is possible that there was a decrease in nitrifying bacteria activity, but at the same time an increase in the activity of other microorganisms, resulting in the insignificant increase in total CO_2 emissions. The increased retention of NH_4^+ would likely lead to increased microbial activity (Wardle et al., 2008), which would offset the decreased activity of the nitrifying bacteria. Although further research is needed into the exact fate of this nitrate, the insignificant change in PP in this study suggests that the decreased nitrate leaching, whatever its cause, has not limited grass production. This goes further towards favouring the NH_4^+ adsorption hypothesis over nitrate immobilisation (as nitrate immobilisation would have reduced PP). It is possible that an increased sorption of NH_4^+ would have promoted PP, but that a simultaneous reduction in NO_3^- production is responsible for the lack of a significant change in PP. Charcoal addition in this study has therefore had a positive effect in terms of nitrate (reduced water pollution), and until any negative effects are found the fate of nitrate should not be used as a reason to avoid biochar application. Nitrate leaching is not only harmful to human health if the high concentrations make their way into drinking water, but it can also result in fish poisoning and algal blooms (Nieder and Benbi 2008), therefore the extent to which nitrate leachate concentration has been reduced in this study suggests that benefits can be gained.

Unlike some other studies (Lehmann et al., 2003; Chan et al., 2007; Major et al., 2009 and Steiner et al., 2007), this research has found no significant change in PP with charcoal treatment. This however is not necessarily a negative, as only a decrease in PP with treatment should discourage the application of biochar, especially in light of reduced nitrate leaching from charcoal application and the benefits it can bring.

In relation to soil water holding capacity, although an improvement in water holding capacity would be the ideal situation in times of drought, the fact that there has been no significant change should not discourage application when the other advantages associated with charcoal addition are considered.

An increase in pH found in this study corresponds to the results of other research (Major et al., 2005). A main benefit of a rise in pH is the economic benefits resulting if biochar can be used as a substitute for lime, and the subsequent increased PP that may result (Knicker 2007), however there are many other benefits associated with a rise in pH which should be considered. Phosphorus availability has been found to increase with increasing pH, and aluminium toxicity in streams is more harmful at a lower pH (Anderson et al. 2000). Although this suggests that the liming effect of biochar is very advantageous, it is important to consider research which has found an increase in DOC leaching as a result of lime application and the subsequent rise in pH (Anderson et al., 2000). This could also help explain the DOC increase found in this study, however pH was not found to have a statistically significant affect on DOC, and as shown in Table 7.4, this negative does not outweigh the other benefits of biochar application, as the DOC losses are tiny in comparison to the carbon sink. The environmental concerns mentioned in relation to DOC must however remain.

As explained above, the charcoal treatment levels used in this research were calculated based on the amounts of carbon that would need to be applied to make the estate used in the study carbon neutral, and that a major aim was to establish whether application could be carried out as a one-off, making the estate carbon neutral for many years (Table 7.2), or whether negative affects at such high cumulative application levels mean that biochar could be applied successfully for a small number of years, but then must stop before negative environmental effects occur. When assessing ideal application rates however, other considerations need to be made relating to the supply rate of biochar if a one-off application is favoured, or alternatively whether application at such low yearly levels will be sufficient enough in terms of PP or environmental related issues to encourage people to take up biochar application.

In relation to pH, statistically significant increases in pH will not occur until treatment level 2 is applied (Table 7.3); however these increases are also present with application of treatment level 3. This suggests that if the advantages of liming are considered a main reason for the uptake of biochar incorporation, then application should proceed as a one-off process with treatment level 3, to achieve this benefit at the earliest possible time. In relation to nitrate leaching, results from the bare arable plots suggest significant decreases even at the lowest treatment level (Table 7.3), and therefore if supplies of biochar are low, benefits will still be gained in the early years of yearly application rates, and that there is no specific need to apply the maximum application rates. Results from the vegetated arable plots and forest soil however suggest that there will be no significant decrease unless the maximum treatment level is applied. This therefore implies that if water quality improvements are seen as an important issue to be solved by biochar application then the maximum treatment level should be applied as a one off application, rather than on a yearly basis, in order to achieve the maximum possible benefits at the earliest time. The results of the PP and biomass measurements showing insignificant change at any application rate mean that neither treatment level should be favoured in order to increase crop yields, and therefore decisions should be guided by the treatment levels most successful for the other factors in Table 7.3. In relation to DOC, increased concentrations were found to be leached from vegetated arable soils with treatment level 2. If the assumption is made that all arable soil will have some sort of vegetation growing, then the implication is that biochar should be applied at treatment level 1 for no longer than 18 years, or as a one off application of no greater than 100 000 Kg/ha (Table 7.3). In relation to the forest soils however it appears that application can proceed at the highest treatment level, if the nitrate leaching and pH benefits are considered of high importance and there is no limit on biochar supply. The same could be said of the arable soils if the benefits of reduced nitrate leaching and pH are weighed up against the small increases in DOC when the net carbon sink is considered (Table 7.4). When considering suitable application rates to apply based on CO_2 respiration data the highest treatment level could be applied as either a one-off application or continuously over the 26 year period on arable soils planted with crops as there was found to be no significant change with any treatment. The data from the bare arable soil however suggests that there would be a significant increase between 1.8 and 18 years of treatment, but that this increase will no longer occur at the highest treatment level. This therefore suggests that the greatest advantages would be gained if the treatment is applied as a one-off process of 140 000 Kg/ha, to avoid any negative affects which could occur at these lower application rates. The same is true of application to forest soils, where the maximum level will achieve the maximum benefits in terms of a carbon sink at the earliest possible time, as well as avoiding any possible increased emissions that could occur at lower application rates (Table 7.3). As discussed earlier, in all situations relating to CO_2 and DOC however, the carbon store decreases that would occur with certain application levels are only minor when the net carbon sink is considered (Table 7.4), and therefore application levels and frequency of application should proceed based on those which are most beneficial for the other factors discussed- nitrate leaching and pH.

Taking all of these factors into consideration and combining the results form bare arable soils and vegetated arable soils to make a suggestion of ideal application rates for arable soils in general, table 4 shows that in 15 out of the 17 carbon flux/parameter changes associated with char application, the benefits to the carbon flux or parameter in question correspond with the highest charcoal treatment level. The following application frequencies and rates are therefore suggested: arable soils: treatment level 3, one-off application; forest soils: treatment level 3, one-off application.

A major aim of this study was to assess the potential of biochar application to more organic rich soils, as well as the more frequently studied low organic carbon mineral soils typical of arable land. This study has shown that the organic rich soils typical of the Wallington estate have behaved similarly to the mineral soils, and that although the sizes of the fluxes of both carbon and nitrogen may differ considerably between the two soil types, the direction of the flux response is similar in both soils. This supports the application to organo-mineral soils on the Wallington estate and means that application is not limited to the small area of arable land located on the estate.

Although this research is an improvement on other studies that have only looked at one aspect of the carbon balance, or specifically at agronomic benefits, it is realised that it has not addressed any changes in CH_4 and N_2O emissions (2 potentially major GHGs) (Richardson et al., 2009), and therefore further research should concentrate on the impact of biochar/charcoal addition on these GHG emissions, to better establish a full GHG budget relating to biochar addition. Recent research from other studies does however suggest that N_2O and CH_4 emissions will also be reduced.(Spokas et al 2009). Another limitation of this study relates to the use of lumpwood charcoal as a substitute for biochar. If biochar addition to soil is to be implemented on a large scale by The National Trust, then the biochar used is likely to be that produced during energy generation on their estates (bringing a further benefit in terms of reduced carbon emissions from avoided fossil fuel use), using animal and farm wastes as a fuel source. It would therefore be beneficial to carry out further research to ensure that this form of biochar behaves in a very similar way to the lumpwood charcoal used here. Although other research suggests that different biochar production mechanisms and materials could have very different effects in soil (Gundale and DeLuca 2007), it is unlikely that the carbon sink benefits found here could be reversed to a carbon source, due to the huge difference in magnitude of carbon sinks versus carbon emissions, even when char was found to result in increased carbon emissions in this study. It is unlikely that a different biochar production method or source would result in an increase in carbon emissions of a level needed to transform the results of this research into a carbon source. The impacts of different types of biochar and production methods on PP and nitrate leaching do however need to be further investigated, as even though carbon sink benefits are likely to remain, any negative impacts on these issues could be a strong case against the uptake of biochar use on an agricultural estate where environmental issues and crop productivity are major areas of concern

7.5 Conclusion

The application of lump-wood charcoal to both a mineral soil and organo-mineral soil typical of north-east England has shown general increases in CO_2 respiration from soil with an increase in charcoal amount; however these increases were very small when compared to the levels of charcoal incorporated into the soil, and they were often insignificant. Accompanied with this increase has also been a significant increased DOC loss from the soil, however again this is small in comparison to the carbon additions to the soil. The results from this chapter show that:

- When the increased losses from NER and DOC when charcoal is applied are accounted for, the soils remain a large carbon sink.
- Significant improvements in nitrate leaching have occurred with charcoal addition, as have significant increases in soil pH: outweighing the increased losses of carbon, as they can bring many environmental, water quality and financial benefits.
- Although there has been no significant increase in PP or water holding capacity, neither has there been a significant decrease, therefore this should not be used as a reason to discourage biochar application, especially when the aforementioned benefits are considered.

Whilst the debate continues as to the reason for these increases in CO_2 loss and DOC loss, this study has shown that whatever the reasons, the carbon sink benefits are large, and that application to both arable and forest soils should be considered in soils similar to those in this study.

Chapter 8

Replacing arable land with short rotation coppice willow: the impacts on SOC and NEE

8.1 Introduction

The impacts on %SOC of changing land-use and land-management have been discussed throughout this thesis, in relation to land-use change out of and into categories of land-use already in existence on the NT's Wallington Estate. In regard to land-use, the %SOC under arable land has been found to be lower than that of any other land category (Chapter 2), and for maximum soil carbon sequestration conversions out of this land-use have been recommended (Chapter 6; Ostle et al., 2009). Land use conversions from arable to improved pasture, arable to rough pasture, arable to woodland, and arable to plantation forestry are all scenarios that will result in SOC sequestration (Chapter 6). In relation to land-management change this thesis has also investigated the impacts of arable fertilisation techniques, cropping systems, and the incorporation of biochar as methods to increase %SOC in arable land. This chapter expands the possibilities of land-use change to look at the effects of a change out of arable land into SRC willow. Investigation of this land-use change is deemed important as SRC willow could be a vital source of biochar if biochar application to land proceeds (Chapter 7), and SRC is also a land-use which has been little studied in relation to its impact on %SOC. With increasing pressure to grow SRC for bioenergy production in recent years (Brandao et al., 2009), and the likelihood that that this pressure may increase in the future, the need for assessment of the impacts on SOC is vital and timely. This chapter will look specifically at the impacts on SOC of converting land from arable crops to SRC willow, but will also assess the impacts on NEE in line with the aims set out in Section 1.3, to measure the impacts of land-use change on both the below and above-ground carbon stock, i.e. the total land carbon stock. This chapter will help guide the

NT as to whether the conversion of arable land to SRC willow can provide a new carbon sink, and fulfil the aims of Section 1.1.1.

8.1.1 SRC, bioenergy crops and SOC: a review of the literature

According to Grogan and Matthews (2001), a form of bioenergy crop, SRC, has been identified by some as the UK's biggest potential carbon mitigation strategy, but a very poor understanding and evidence base regarding its impact on SOC makes the need for further research vital (Ostle et al., 2009). SRC willow currently covers 2600 ha of land in the UK, and is a fast growing perennial woody crop harvested every three years, and used as a fuel for power generation (Brandao et al., 2010). Many previous studies looking at the impacts of SRC and bioenergy crops on the land carbon balance have had a tendency to ignore the impact on SOC, and although there is a large volume of work comparing forest and woodland SOC stocks with those of pasture and arable lands (e.g. Bolstad and Vose, 2005), much of this is not specific to bio-energy woodland, where the impacts on SOC could be very different to natural woodland.

The majority of research into bioenergy crop plantation has focused on the gains in carbon achievable through sequestration into bio-crop tree biomass, however the impact of plantations on SOC is little studied (Brandao et al., 2010). Although the atmospheric carbon sequestered into bioenergy crop and tree biomass is likely to be significantly greater than that sequestered into arable crops or the grassland which it is to replace, this can not be used as a reason to convert to this land-use, until the impact on the SOC store upon conversion is confirmed. There is general agreement in the scientific community that a much more thorough knowledge of the impact of such plantations on soils is needed before land conversion of this type can proceed (Jug et al., 1999).

Although the impact of bioenergy plantations on SOC is under-studied, there are some results that suggest the likely outcomes of such plantations, but many of these are conflicting, and come from areas of the world where soil and climate conditions differ significantly from those of the UK. The term bioenergy crop also refers to a wide variety of species, and to therefore assume that the impact of each species on SOC will be the same could be very inaccurate. A study by Anderson–Teixeira et al (2009), for example, states that the impact of bioenergy crop plantation on SOC stocks is variable; however none of the studies that they assess are associated with the plantation of SRC willow. If the plantation of SRC willow is to proceed in north east England there is a need for much greater research, into not only the impact of bioenergy crops on SOC, but also into the impact of particular species (in this instance SRC willow), in specific climatic and soil conditions, on SOC. Much of the evidence behind the claim that SRC can be the UK's greatest potential mitigation strategy, comes from only one piece of research on natural woodland regeneration (Grogan and Matthews, 2001), further emphasising the need for research specific to specific bioenergy crops. Not only are the tree species associated with natural woodland regeneration very different to bio-energy crops, but the harvesting and management procedures undertaken in bioenergy plantations will not be accounted for in any assessment of the impacts of natural woodland on SOC (Grogan and Matthews, 2001).

Although several studies have found a loss of SOC following plantation establishment (Anderson – Teixeira et al., 2009), it is possible that this is the result of land conversion loss, and the losses of SOC associated with the land-use lying fallow for a period before plantation establishment, or losses due to tillage and the ploughing of land necessary for tree plantation (Tolbert et al., 2002). Jug et al (1999) observed increased SOC losses following conversion from arable to bio-energy crop plantation, however these were attributed to the decomposition of harvest and litter residues remaining from the former land-use. These losses may thus only be short-lived, and to assume that the bioenergy plantation has caused declines in SOC stocks would be inaccurate if the findings were based only on the result of samples taken in the immediate years following establishment. Other studies, however, suggest that the combination of below-ground and above-ground surface NER from a bioenergy plantation could be much greater than that from an arable or pastoral land-use due to tree root and litter respiration (Gordon et al., 2005), implying that the greater losses of carbon to the atmosphere under bioenergy crops may be a more permanent feature.

These conflicting results and outcomes relative to different locations are revealed when different studies are compared. A study in Ireland looking at the plantation of miscanthus did not find decreases in %SOC on former arable land following plantation, as in the studies above, but increases (Dondini et al., 2010). These results are supported by a study in the United States (Coleman et al, 2004), which found SOC stocks under some areas of a SRC plantation to be higher than those under neighbouring arable land. This study however, also highlights the variability in %SOC with such a land-use change, and the inability to reach firm conclusions regarding the impact of plantation establishment, with some other areas of the SRC plantation found to have lower %SOC.

8.1.2 Chapter aims

The aim of this chapter was to establish the impact of SRC willow plantation on the total land carbon balance when planted on former arable land in north east England. The aim was to assess whether any carbon gains could be achieved with a conversion from traditional arable land to the growth of bio-energy crops. Comparison with arable land was made as this was believed to be the land-use with the greatest likelihood of conversion to SRC. This belief stemmed from the fact that %SOC under arable land is generally lower than that under pasture and woodland (Bell and Worrall, 2009). This land-use change is hence the most likely as it will minimise any losses if plantation results in a release of SOC, or conversely maximise any gains if plantation results in an increase in SOC.

A further objective was to attempt to uncover some of the issues related to land-conversion losses of SOC, and the possibility of SOC gain/loss reversal following several years of bioenergy crop plantation establishment as discussed in Section 8.1.1. It was therefore decided to undertake the trial on a site consisting of two different aged plantations, to help clarify the variation in SOC stock with plantation age, and hence be indicative of any land conversion loss.

The approach taken by this chapter was to use a variety of field methods and literature studies so that the impacts on both below and above-ground carbon can be identified. Due to the incompatibility in size of the soil chambers used to measure NEE and PP, and the size of vegetation under the two SRC plantations, NEE and PP could not be measured directly in this study. It was therefore decided to monitor NER from the soil surface (SOC losses) under each land-use, and to identify SOC gains by collecting litter fall from beneath each land-use. Literature studies would then be undertaken to identify the carbon gained in the above-ground vegetation of each land-use as PP. These gains in PP would then be combined with the losses/gains in SOC under each land-use to reveal the net gain/loss in the combined above and below-ground carbon stocks.

In addition to the measurement of carbon fluxes, it was also aimed to directly measure %SOC under each land-use, and to assess whether any apparent changes due to land-use could be identified after only a short number of years of land-use implementation.

8.2 Materials and Methods

8.2.1 Study site

The study took place at Newcastle University Cockle Park farm in Northumberland, north east England (Figure 8.1). This study site was chosen due to its proximate location to the NT Wallington Estate (Figure 1.1) meaning that climate conditions, soil types and other variables likely to have an impact on %SOC (Section 2.1.2) were as consistent as possible between the trial and Wallington site.

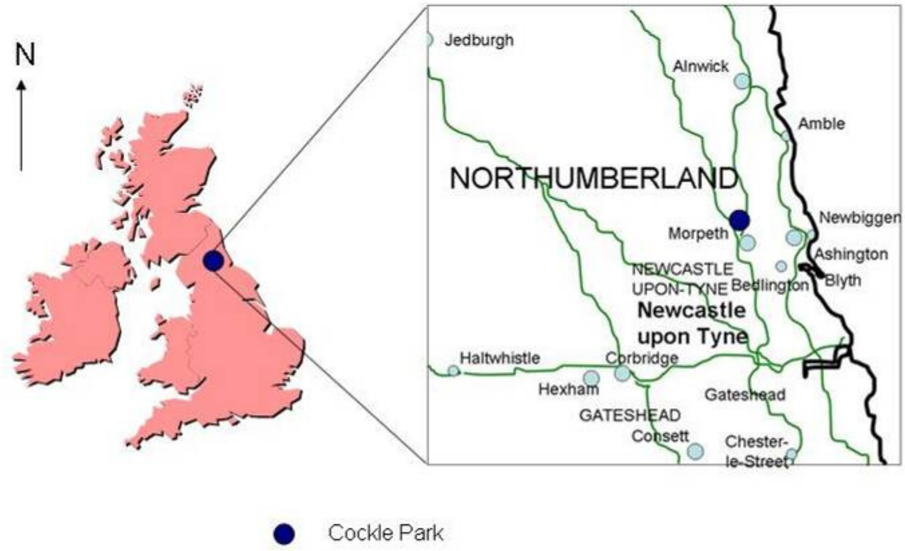

Figure 8.1 Study site location: Cockle Park Farm, Northumberland

The study was not undertaken on the Wallington Estate as there is currently no land under SRC. Although a SRC plantation could have been created, and the carbon fluxes and SOC stocks following one year of plantation growth compared to that of arable land, this would only identify the short term impacts and not meet the required aims of this chapter.

The study compares the %SOCs, carbon fluxes and above-ground carbon stocks of three areas of land at Cockle Park: an arable field (control site), a SRC plantation planted in 2006 (henceforward known as SRC 2006) and a SRC plantation planted in 2008 (henceforward known as SRC 2008). The variation in these land-uses upon trial implementation can be seen in Figure 8.2 a-c. This specific site was chosen for investigation as all three land-types under investigation were located on the same soil type (disturbed soils- former opencast mine site (Payton and Palmer, 1989)), spanned the same range in altitude, and had a similar topography and land-use history. This meant that the effect of current land-use could be studied in isolation, without the risk of being masked by other variables.

Figure 8.2 a. Arable field **Figure 8.2 b. SRC 2006** **Figure 8.2 c. SRC 2008**

The study ran for a total period of one year, over which time significant growth in both SRC plantations was apparent. The change in appearance of SRC 2006 can be seen by comparison of Figure 8.2b and Figure 8.3.

Figure 8.3 SRC 2006 following one year of monitoring

The arable field in this study has been in a continuous arable rotation system for 12 years, since 1997, when it was planted on a former opencast mine site. Prior to 2005 both areas of land now under SRC plantation had the same land-use history as the arable field. The arable rotation system consisted of one year Winter Wheat, one year set-aside, one year Spring Barley, one year Winter Barley, one year Oilseed Rape and two years Winter Wheat. SRC 2006 was planted in April 2006, following a brief period of fallow. SRC 2006 covers a land area of 4 hectares and was planted with a mixture of 13 Willow varieties in double rows. This resulted in 13 lots of 3 double rows. The rows were planted at 75 cm intervals, with 150 cm separating each double row. SRC 2008 also covers a land area of 4 hectares, and was planted as a completely randomly mixed plantation with no distinction made for willow variety. Prior to plantation, the land now planted as SRC 2008 had lain fallow for 2 years.

The method used in this study was a space-for-time substitution method, where it was assumed that the previous land-use history and SOC values were consistent for all current land-use areas before they were put into their current land-use. For the control an adjacent field was chosen, where the land-use matched that of the SRC plantation pre-establishment land-use. Although this method is often criticised (Williams et al., 2008), it was considered the best available, as the time period of study was limited, and to assess SOC stocks on the same piece of land both before and after a land-use change would take many years (Heim et al., 2009).

8.2.2 %SOC variation with land-use and change over time

As with the other land-use intervention trials undertaken in this thesis (Chapter 4, Chapter 9) it was initially thought unlikely that a difference in %SOC would be evident following only a short number of years of current land-use implementation (Williams et al., 2008). It was, however, still considered important to investigate whether any difference in %SOC was evident between land-uses, and so samples were collected and compared for %SOC from beneath each land-use prior to the start of the trial. Samples were taken in October 2008, before any NER measurements were made. These measurements provided an average %SOC value for long term arable land, a SRC plantation less than 6 months old (SRC 2008) and a SRC plantation of 2.5 years (SRC 2006). The large spatial variation often found in %SOC (Saby et al., 2008; Tolbert et al., 2002) meant that a high sampling density was required. A total of 30 samples were therefore taken from beneath each land-use. A further set of 10 samples were taken from beneath each land-use at the end of the trial, when the current land-uses had been in existence for a further year. Again this was done to see if any significant differences were apparent between land-uses, although it was realised that due to the time that SOC needs to adjust to land-use change it was unlikely that any changes would be detectable.

For each sample taken in the field a GPS location was recorded and notes of the altitude, aspect and land-use made. Any relevant notes on landscape position (e.g. topographic decline) were also taken as this is recognised to control %SOC (Dick and Gregorich, 2004). The land-use at each sample point was identified as either arable, SRC 2006 or SRC 2008. It was recognised that any soil samples taken would need to be accurate representations of the area from which they were taken in order to provide reliable results (Cook and Elis, 1987), therefore a stratified random sampling technique was adopted to break down each land-area into a number of subareas and then a random sample taken from each. Stratification was based on topography and slope aspect. Samples from areas close to field boundaries were avoided due to the possibilities of compaction from machinery resulting in an unrepresentative sample, as were the corners of fields (which may have been sites for crop and fertiliser storage), gate entrances and other unrepresentative areas. Although SRC 2006 consisted of 13 willow varieties in uniform rows it was decided not to assess the possibility of differences in %SOC caused by willow variety, therefore samples were taken at random with no consideration of SRC tree species. The reasoning behind this was that there was no distinction between varieties within SRC 2008, and to therefore add this extra variable into only one of the land-uses would make the controls on NER difficult to establish.

Measurements of %SOC were made by collecting a sample from a depth of 20 cm using an auger (the depth to which SOC in mineral soils is most likely to be affected by land-use change), (Woomer et al., 2001; Cheng and Kimble, 2001; Kimble et al., 2001). All samples were placed in sample bags in the field, labelled and transported back to the laboratory where they were dried overnight at 105°C and stored. LOI and the Walkley-Black wet oxidation method (De Vos et al., 2007) were used to establish the %SOC of each sample as described in Section 2.2.2.1, with triplicate or duplicate measurements made on each.

8.2.3 Ground surface NER

All NER measurements made in this study refer to respiration from the ground surface and do not include any respiration from either arable-crop or SRC willow above-ground biomass. The measurements will however include respiration from below-ground arable crop and SRC willow roots, weeds and grass.

6 PVC collars were inserted into the ground under SRC 2006, 6 into the ground under SRC 2008 and 6 into the ground under the arable field in October 2008. As with the soil samples taken for measurement of %SOC the collars were located using a stratified random sampling technique within each land-use. Under SRC 2006 again no stratification for willow variety was made. The collars were inserted to a depth of approximately 5 cm, leaving 5 cm protruding from the surface on which to place the soil respiration chambers (Figure 8.4). These collars were left in place for two weeks before any measurements were

made to ensure that measurements of any of the effects of root death caused by insertion of the collar would be minimised. The collars remained permanently located in the ground for the duration of the trial.

Figure 8.4 The installation of soil respiration collars beneath each land-use

NER was monitored on a fortnightly basis from the ground surface using a dynamic dark closed chamber and Infra-red gas analyser, following the methodology described in Section 4.2.4.1, allowing the change in CO_2 concentration within the chamber to be measured, and hence the CO_2 efflux from the soil surface. Readings were taken between the hours of 10am and 12.30pm on the first visit, and every visit following in order to minimise the effects of any diurnal changes in NER which may obscure other factors responsible for differences in NER. Surface-air temperature was recorded by a temperature probe located within the soil chamber alongside each measurement of NER, to allow the NER flux to be calculated as in Section 4.2.4.1, and so that surface-air temperature could be included as a variable in any analysis assessing the controls on NER.

8.2.4 Leaf litter collection

As indicated in Section 8.2.1 measurement of the gains in SOC were needed in addition to the measurement of losses as NER. Litter trays were therefore put in place alongside each collar under each of the three land-uses (Figure 8.5). These litter trays would measure the carbon accumulated in leaf litter, some of which would go on to be converted to SOC, although it was also recognised that some would be released as CO_2. Two trays with small holes for drainage were secured to the ground next to each collar and left to collect litter. Litter accumulation was monitored on each site visit and collected following significant accumulation. The litter accumulated from each plot was placed into labelled sample bags and transported back to the laboratory. In the laboratory it was dried in an oven at 70°C for 48 hours following the method of Cortrufo et al (2005). The litter was then weighed and the weight recorded. This same procedure was followed and litter collected every time sufficient litter to be collected had accumulated. The direct weight of litter was recorded and then converted to weight of carbon, assuming a leaf litter carbon content of 50% (Singh and Lodhyial, 2009).

Figure 8.5 Litter trays to measure carbon accumulated in leaf litter

8.2.5 Biomass carbon gains

The amount of carbon sequestered in the biomass of each land-use would ideally have been measured directly by monitoring NER and NEE and calculating PP as in Section 4.2.4.3. As explained in Section 8.1.2 however, the size and height of the SRC willow biomass meant that the soil respiration chambers used for monitoring NER and NEE of vegetation could not be used. A literature review was therefore undertaken to estimate the mean amounts of carbon sequestered per year into the above-ground biomass of SRC willow bio-energy crops. Carbon sequestration into arable above-ground biomass was estimated by collecting a sample of the crop from a known area following crop harvest, but before removal of the harvest from the field. This biomass was collected from the field and transported back to the laboratory in sample bags. As with the leaf litter it was dried for 48 hours in an oven at 70 °C and its weight recorded. As this arable crop was an annual crop the carbon value refers to the total amount of carbon sequestered into the above-ground biomass of arable crops per year. Where carbon values from the literature were presented as both weights of dry matter and weights of carbon the given values of carbon were recorded. In situations where only the weight of dry matter was provided this was converted to carbon by assuming a carbon content of dry matter of 50% (Singh and Ladhyial, 2009). The same carbon content was assumed for the arable harvest collected from Cockle Park. These values of biomass carbon sequestration rates represent net PP, and have therefore already taken account of the losses to the atmosphere as respiration.

8.2.6 Statistical analysis

All analysis of NER and %SOC was performed using MINITAB 14 statistical analysis software. Comparison of %SOC between land-uses prior to the start of the trial was done by one-way ANOVA as the nature of the study site meant that all other variables were controlled (see Section 8.2.1). The same statistical analysis was used to compare %SOC between land-uses at the end of the trial. Analysis of NER between land-uses was undertaken using GLMs, and ANOVA and ANCOVA were used to establish if these land-use controls remained significant with uncontrolled experimental variables held constant. The factors investigated in this study as controls on NER were land-use and month of measurement. A variation in surface-air temperature with measurement meant that surface-air temperature was entered as a covariate in all analyses. Regression analysis of surface-air temperature and NER was undertaken to enable annual NER fluxes to be calculated for each respective land-use. A normality test was undertaken to assess the distribution of each variable, and if the distribution became more normal upon log-transformation the log-transformed value was used in all analysis. Any significant difference identified by ANCOVA or ANOVA was then post-hoc tested using the Tukey test, to identify between which factors significant differences in %SOC or NER occurred. The r^2 values generated by ANCOVA represent the between *group sum of squares* divided by the *total sum of squares,* with a large r^2 value thus indicating that a large fraction of the variation in the independent variable can be explained by the categorical variable/treatment. The r^2 value represents the proportion of the total variation explained by the difference in the means.

8.2.7 Annual NER flux calculations

The NER fluxes (g C/m^2/hr) recorded in this study were calculated from measurements made between the hours of 10am and 12.30pm. These calculated hourly fluxes can not be assumed to be representative of mean daily fluxes as NER is affected by surface-air temperature (Figure 8.8; Lloyd and Taylor, 1994), and the temperature between 10am – 12.30pm is likely to be unrepresentative of the mean daily temperature due to diurnal variation. When investigating the impact of land-use on NER the calculated hourly fluxes could be compared, as all land-use NER fluxes were measured over the same time

period, however when calculating annual fluxes the impact of temperature must be taken into account. It was therefore considered that an accurate estimate of annual NER flux would not be achieved by simple multiplication of the mean recorded NER flux for each land-use by the number of hours in a year, but instead by obtaining an equation for the relationship between temperature and NER, and calculating annual NER flux using mean daily temperatures for the each month of the year. A mean monthly temperature was therefore required, and was obtained from the Durham University weather station (www.geography.dur.ac.uk/projects/weather/Home/tabid/666/Default.aspx).
These temperatures are displayed in Table 8.1. A lack of average monthly temperatures specific to the Cockle Park site meant that the nearest accessible and available weather station data were utilised.

Month	Mean monthly temperature (°C)
January	3.4
February	4.7
March	6.8
April	8.9
May	11.3
June	13.5
July	15.4
August	15.8
September	13.6
October	9.0
November	6.5
December	3.6

Table 8.1 The mean monthly temperatures for Durham City

8.3 Results

All NER and surface-air temperature measurements made in this trial can be found in Appendix 8 under the heading ‘CO2 flux data’. All pre and post-trial %SOC data is located in Appendix 8 under the headings ‘Initial %SOC’ and ‘Final %SOC’, and leaf litter data under the heading ‘leaf litter’.

8.3.1 %SOC variation with land-use

8.3.1.1 Initial %SOC variation with land-use

The lowest %SOC was found below SRC 2006, with a mean %SOC of 1.61%. The control site (arable field) had a mean %SOC of 2.02%, with the highest %SOC of 2.37% found below SRC 2008. Although the difference in %SOC between land-uses was statistically significant, the variation in land-use could only explain 19.23% of the variation in %SOC. Post-hoc analysis of the results revealed that the difference was only significant between SRC 2006 and SRC 2008. With only 19.23% of the variation explained by the variation in land-use other variables not measured in this study must be responsible for a large amount of the variation in %SOC.

8.3.1.2 Post trial variations in %SOC with land-use

The lowest %SOC one year after the initial measurements were taken was found below SRC 2006, with a mean %SOC of 1.73%. The control site (arable field) had a mean %SOC of 1.82%, and the highest %SOC was again found below the 2008 SRC plantation, with a mean %SOC of 2.44%. Although the order of %SOC is the same as that prior to the trial, with %SOC decreasing in the order SRC 2008 > arable > SRC 2006, these differences are no longer statistically significant. This however may be due to the smaller sample size, with only 10 %SOC measurements made below each land-use post-trial compared to a sample size of 30, pre-trial.

8.3.2 Leaf litter and biomass gains

8.3.2.1 Leaf litter

There was a statistically significant difference in leaf litter between land-uses, with the greatest litter production under SRC 2006, and the lowest in the arable field where no leaf litter was produced (Figure 8.6). The total annual litter fall presented in Figure 8.6 represents the mean total dried weight of the vegetation collected from the litter trays displayed in Figure 8.5.

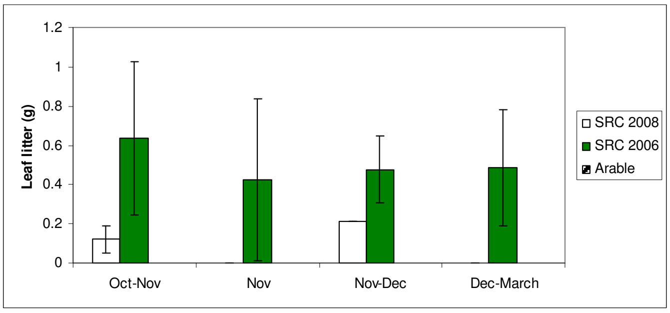

Figure 8.6 The variation in total mean annual leaf litter fall with land-use. Error bars indicate the standard deviation from the mean.

Assuming a carbon content for SRC Willow of 50% based on typical vegetation carbon contents (Singh and Lodhyial, 2009), the total annual leaf litter accumulated beneath SRC 2006 was calculated as 0.69 t C/ha/yr, below SRC 2008 as 0.21 t C/ha/yr and below arable land use of 0 t C/ha/yr. Assuming a carbon content of 50% in this study was thought sufficient due to time and equipment limitations making direct measurement of biomass carbon content difficult. This assumption must however be taken with caution due to the variation in vegetation carbon content depending on the species (Ho, 1976).

8.3.2.2 Above-ground biomass

The results of a literature review undertaken to provide an estimate of the above-ground carbon stocks of a typical SRC willow plantation and arable field in north east England are presented in Tables 8.2 and 8.3.

Author	Above-ground biomass carbon sequestration rate (t C/ha/yr)
Grogan and Matthews (2001)	6
Rowe et al (2008)	4.25
Heller et al (2003)	5
Forest research (2010)	9
Brandao et al (2010)	7.38
Borzecka-Walker et al (2008)	6.25
Fischer et al (2005)	7.98
Mean	6.55

Table 8.2 Literature review of typical above-ground SRC biomass carbon sequestration rates

Author	Above-ground biomass carbon sequestration rate (t C ha/yr)
Adger and Subak (1996)	2.2
Dahl and Anderson (2007)	1.9
Tomlinson (2006)	3.2
Milne and Smith	1
Falloon et al (2004)	2.2
Ordonez et al (2008)	0.5 - 9
Milne and Brown (1997)	1
This study	2.65
Mean	2.97

Table 8.3 Literature review of typical above-ground arable biomass carbon sequestration rates

The results of the review into SRC above-ground biomass reveal a mean annual carbon sequestration rate of 6.55 t C/ha. The review into typical arable above-ground carbon stocks and the carbon stock of the arable crop harvest obtained from Cockle Park is presented in Table 8.3. The total biomass collected from each of the six plots following arable crop harvest at Cockle Park was averaged, and a value of 2.65 t C/ha calculated for annual PP of arable crop above-ground biomass. This value compared favourably to values taken from a review of the literature, providing a mean value of 2.97 t C/ha/yr for above-ground carbon gains as PP in arable vegetation.

8.3.3 Surface-air temperature

Surface-air temperature measurements made alongside each measurement of NER revealed, as expected, a seasonal temperature variation as shown in Figure 8.7.

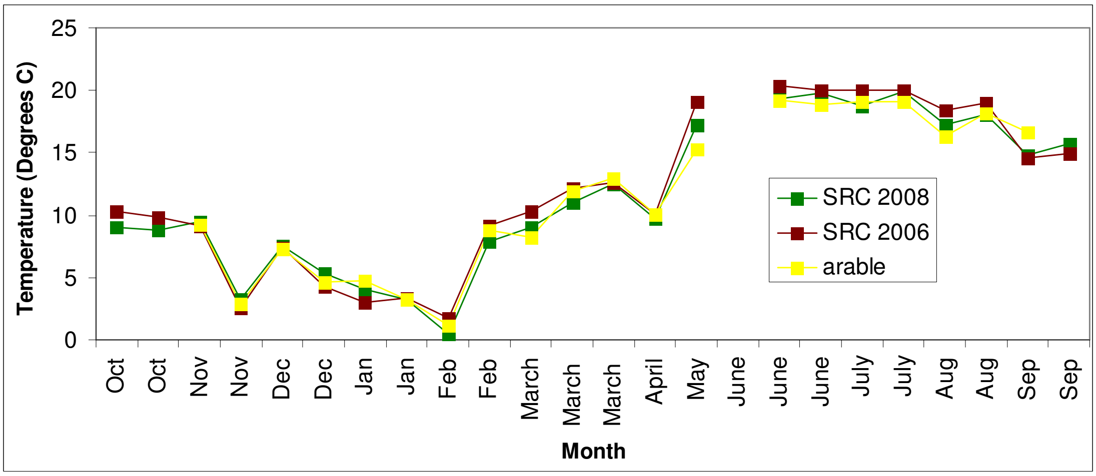

Figure 8.7 Variation in mean surface-air temperature over the trial period

It can be seen however that there was no significant variation in temperature with land-use. As explained in Section 8.2.3 these temperature measurements were taken between the hours of 10am and 12.30pm and are therefore not representative of mean monthly temperatures as they do not take account of night-time temperature.

8.3.4 Controls on NER

NER measurements taken over the 12 month period resulted in a total of 138 measurements from SRC 2006, 143 from SRC 2008, and 124 from the arable land-use. These consisted of 24 fortnightly readings from each of the six sample locations under each land-use respectively, with slightly less from the arable land and SRC 2006 due to equipment failure or crop harvest. The fewer readings from the arable field were the result of harvesting the arable crop at the start and end of the trial, meaning that the soil respiration collars had to be removed from the field. One way ANOVA revealed that over the entire measurement period there was a statistically significant difference in NER between land-uses, with the highest mean NER of 0.11 g C/m^2/hr occurring under SRC 2006, followed by a mean NER of 0.09 g C/m^2/hr under SRC 2008, and a mean NER of 0.07 g C/m^2/hr under the arable field. Although this difference was significant, the variation in land-use alone could only explain 2.26% of the variation in NER.

Despite the significant difference in mean NER over the 12 month period with land-use, one-way ANOVA did not reveal how NER varies with time of year, or under which months any significant difference in NER with land-use does occur. The factors month and land-use were therefore entered into a GLM and ANOVA undertaken to reveal that both factors could explain some of the variation in NER. The variation in month and land-use could together explain 78.14% of the variation in NER. The interaction between month and land-use was also entered into the model; however the lack of measurements from the arable land-use during the month October meant that all readings taken in October had to be removed from the dataset. With the interaction between month and land-use included in the GLM, variation in the factors could together explain 81.87% of the variation in NER, with land-use, month and the interaction between month and land-use all being significant controls on NER. The variation in NER with month and land-use over the total trial period is shown in Figure 8.8.

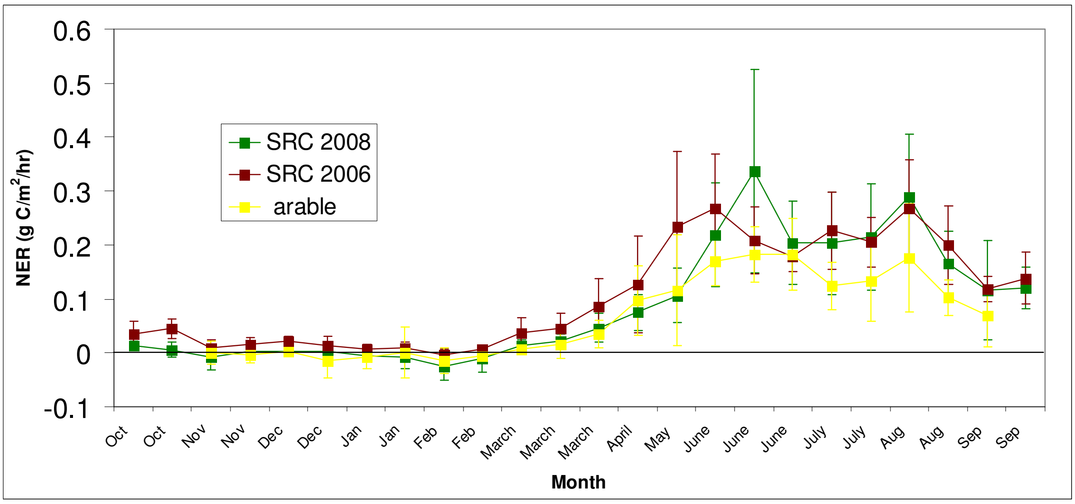

Figure 8.8 The variation in mean NER from the soil with land-use and month of measurement. Error bars represent the standard deviation from the mean.

This interaction between month and land-use indicates that the greatest losses of carbon occur from different land-uses depending on the time of year. Post-hoc analysis of the results revealed that under no individual month was the NER flux significantly different with land-use. The error bars in Figure 8.8 indicate the lack of a significant difference in NER with land-use for individual months, and the interaction between month and land-use can be observed, with the highest NER flux occurring under different land-uses depending on the month of year. In relation to the mean NER from the total trial period, post-hoc analysis revealed a significantly greater loss of carbon from SRC 2006 than from SRC 2008, a significantly greater flux from SRC 2006 than from arable land, and a significantly greater loss of carbon from SRC 2008 than from arable land.

With the covariate surface-air temperature included in the model all other factors remained significant and temperature itself did not have a significant effect on NER, indicating that land-use and month are both significant controls on NER, and that temperature variations beneath different land-uses and months are not responsible for the significant effects caused by these factors.

8.3.4.1 Surface-air temperature and NER

The insignificance of surface-air temperature in this study is the likely result of inclusion of month of measurement in the analysis, and thus colinearity between surface-air temperature and month. To establish the true control of surface-air temperature on NER a regression analysis of surface-air temperature against NER was therefore undertaken. This revealed that 63.4% of the variation in NER could be explained by the variation in temperature. The relationship between temperature and NER can be observed by comparison of Figures 8.7 and 8.8, and is demonstrated further in Figure 8.9, with the regression equation displayed in Equation 8.1.

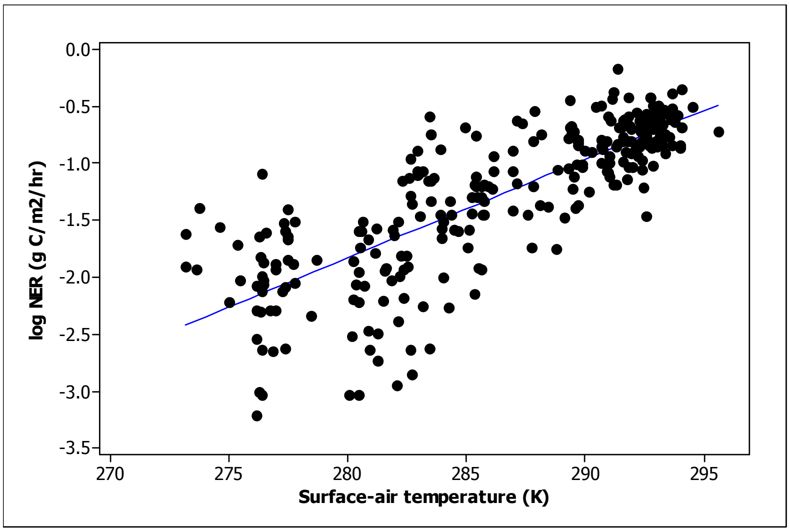

Figure 8.9 The relationship between temperature and NER, where K: degrees Kelvin

$$Log\ NER = -26 + 0.086K$$ **Equation 8.1** r^2 = 63.4%, n = 278

Where: NER: g C/m2/hr; K: temperature (Kelvin)

8.4 Discussion

8.4.1 %SOC and land-use

The results from the %SOC values taken from beneath each land-use prior to the start of the trial suggest that conversion from arable crops to SRC willow bioenergy plantations will initially cause a rise in %SOC, supporting the findings of Dondini et al (2010). SRC 2008 was approximately 6 months old at this time, and this suggestion is based on the findings of higher %SOCs below SRC 2008 than below the arable field. This suggests that atmospheric carbon sequestration into SOC will increase with this land-use change for at least the first six months, adding to the increases in carbon sequestration which are also likely to occur in terms of the greater gains in PP of above-ground biomass under this land-use (Section 8.3.3.2). When SRC 2008 %SOCs are compared with the older of the two plantations (SRC 2006) however, it appears that these gains in %SOC may start to decline several years after plantation establishment. SRC 2006 was approximately 30 months old at this time, and this suggestion is based on the findings of lower %SOC below SRC 2006 than below the arable field. Had only the younger of the plantations been compared to the arable field this decline in %SOC over time would not have been recognised, making this study an improvement on any short-term trials. These results from the samples taken in 2008 suggest that the plantation of SRC willow for bio-energy generation will cause an initial rise in %SOC, and hence sequestration of atmospheric carbon over the first 6 months of plantation establishment, but that SOC sequestration will cease, and SOC emissions will begin to occur somewhere between 6 and 30 months following plantation establishment. It is important to note here, however, that the increases in SOC observed below SRC 2008 may be the result of organic manures added to the soil upon plantation establishment, and not directly as a result of the land-use change. This caution in the interpretation of the results must be taken due to the observance of manure type substances within the soil samples, although no information regarding manure application has been provided. It is also important to note that the observed increases in %SOC were not significantly different between arable land and SRC 2008, therefore the loss of carbon with SRC plantation 30 months after establishment should be considered the more firm conclusion of this trial.

The results from the %SOC values of the samples taken at the end of the trial in 2009 show a similar trend to those taken one year previously, prior to commencement of the trial. This again suggests that there will be initial increases in the amount of carbon sequestered into soil with conversion from arable to SRC willow bioenergy plantations, however that these increases will not be long-lasting, and that SOC will begin to decline, and soil carbon emissions to the atmosphere increase as the plantation ages. The higher %SOCs beneath SRC 2008 compared to SRC 2006 and arable land approximately 18 months after plantation establishment do however help to constrain the age range at which the plantation is likely to cease sequestering carbon as SOC, and convert to being an emitter of SOC. With %SOC still greater under SRC willow 18 months after plantation establishment the earlier quoted figure of transmission from a sink to a source 6 – 30 months following plantation establishment can be adjusted to 18 – 30 months. These findings of greater %SOC beneath SRC 2008 do not agree with those of Anderson-Teixeira et al (2009), who found a loss of SOC due to land conversion loss. Despite this land lying fallow since 2006 it does not appear to have caused a decline in %SOC. None of these differences in %SOC measured in 2009 were statistically significant and should therefore be taken with caution.

8.4.2 NER and land-use

The results of the NER trial indicate a significant difference in NER with land-use, with the greatest losses of SOC from SRC 2006, and the smallest losses from arable land. These findings support those of Gordon et al (2005) who suggest greater NER fluxes from bioenergy plantations due to tree root and litter respiration. Although NER was greater under SRC 2008 than the neighbouring arable land, it is unlikely that this is the result of the decomposition of harvest residues from former arable land-use, as suggested by Jug et al (1999), as this land had been fallow for the previous two years. This significant difference with land-use was true when the mean NER flux from each land-use over the trial period was compared, however under no individual months were the differences with land-use significant. Although surface-air temperature does explain some of the variability in NER it has an insignificant impact when land-use and month are controlled – month and surface-air temperature were probably collinear. This indicates that the variability still unexplained by land-use and month can not be explained by surface-air temperature. The fact that land-use and month, and the land-use/month interaction remained significant with surface-air temperature included in the analysis indicates that land-use has a significant affect on NER, and that the effects of land-use are not the result of land-use causing a difference in temperature. This is further confirmed by reference to Figure 8.7, where it can be seen that surface-air temperature does not vary significantly with land-use.

Although these results suggest that soil NER is greater beneath SRC plantations than arable land, these measurements also include NER from below-ground roots. No measurements in this study have been made of the NEE and PP of the below-ground roots, therefore to suggest that SRC plantation should not proceed due to increased losses from the soil surface would be a false assumption to make until measurement of carbon uptake in SRC roots can be measured, and compared to that of arable crop roots.

8.4.2.1 Calculating the annual NER flux

The results of ANCOVA revealed that land-use has a significant effect on NER. To accurately calculate the annual NER flux from each land-use required the relationship between surface-air temperature and NER to be known for each land-use. Regression analysis of NER against temperature for each land-use produced a regression equation allowing the annual NER flux for each land-use to be predicted using the mean monthly temperatures displayed in Table 8.1.

NER for SRC 2006 was predicted using Equation 8.2, and the relationship between NER and temperature under this land-use is displayed in Figure 8.10.

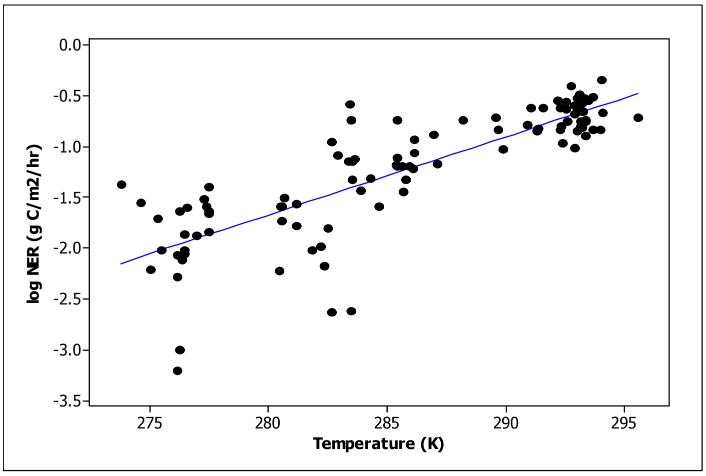

Figure 8.10 The relationship between temperature and NER under SRC 2006

$LogNER \equiv -23.11 + 0.077K$ **Equation 8.2** $r^2 = 67.9\%$; n = 102

Where: NER: (g C/m^2/hr); K: temperature (Kelvin)

NER for SRC 2008 was predicted using Equation 8.3, and the relationship between NER and temperature under this land-use is displayed in Figure 8.11.

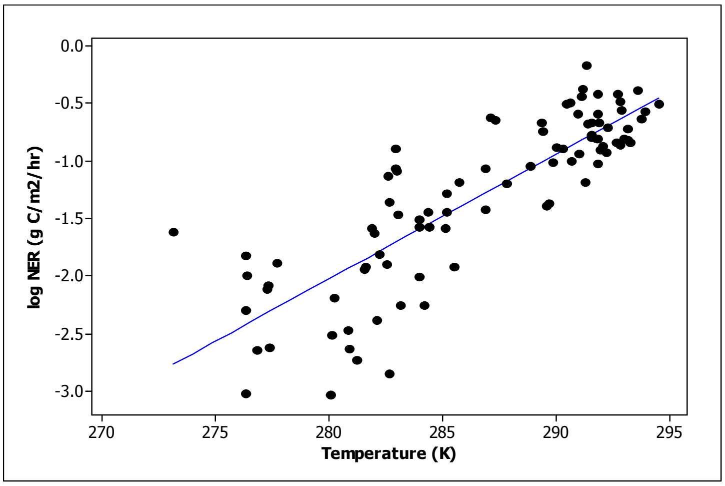

Figure 8.11 The relationship between temperature and NER under SRC 2008

$Log\ NER\ = -32.37 + 0.108K$ **Equation 8.3** $r^2 = 70.1\%$; n = 90

Where: NER: g C/m^2/hr; K: temperature (Kelvin)

NER for arable land was predicted using Equation 8.4, and the relationship between NER and temperature under this land-use is displayed in Figure 8.12.

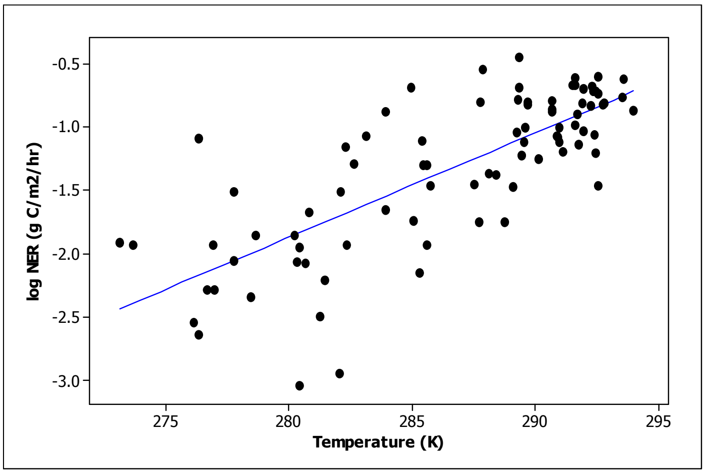

Figure 8.12 The relationship between temperature and NER under arable land

$Log\ NER\ = -24.93 + 0.082K$ **Equation 8.4** $r^2 = 58.4\%$, n = 84

Where: NER: g C/m^2/hr; K: temperature (Kelvin)

NER for each hour of each month was calculated and these hourly values were summed to produce monthly NER values. Addition of these monthly NER values produced an annual estimate of NER for each land-use. These monthly values and annual NER fluxes are presented in Table 8.4.

Month	Arable	g C/m^2/month SRC 2008	SRC 2006
January	5.05	2.91	8.42
February	5.83	3.63	9.57
March	9.62	6.79	15.34
April	13.86	11.10	21.50
May	22.58	20.88	33.92
June	33.16	34.99	48.38
July	49.13	58.09	69.88
August	53.00	64.19	74.99
September	33.79	35.87	49.24
October	14.60	11.76	22.61
November	8.79	6.10	14.08
December	5.24	3.06	8.73
Annual NER (g C/m^2)	254.65	259.37	376.65

Table 8.4 Calculation of annual NER for all land-uses

8.4.3 Land-use and the total land carbon balance

Summation of all of the aforementioned carbon losses/gains under the three different land-uses investigated in this chapter provides an indication of the impact of bioenergy plantations (in particular SRC willow) on the total land carbon balance. The annual carbon gains and losses associated with each land-use in this study, and the impact on the carbon balance are presented in Table 8.5.

Land-use	Carbon lost from soil as NER (t C/ha)	Carbon added to soil in litter (t C/ha)	Carbon gained in crop/tree biomass (t C/ha)	Sink/source of carbon (t C/ha)
Arable	2.55	0	2.97	- 0.42
SRC 2008	2.59	0.212	6.55	- 4.17
SRC 2006	3.77	0.691	6.55	-3.47

Table 8.5 The combined annual affects of land-use. A positive value represents a loss to the atmosphere and a negative value a carbon sink.

The smallest loss of SOC as NER from arable land indicated in Table 8.5 agrees with the %SOC values from both the pre and post-trial soil sampling, with the lowest NER flux corresponding with the greatest %SOCs. Loss of SOC as NER in the order of SRC 2006 >SRC 2008 >arable (Table 8.5) corresponds with the order of %SOC where arable >SRC 2008 >SRC 2006. Although the gains in SOC from leaf litter addition are greatest under SRC 2006 these are not great enough to offset the losses from NER.

Initial observation of Table 8.5 suggests that arable land is a small sink of carbon and that conversion from this land-use to SRC willow will create a larger sink of carbon within the first year of plantation establishment, and that SRC willow plantations will continue to be sinks of carbon after three years of plantation establishment, but to a lesser extent. These conclusions are however reached with necessary measurements of other carbon loss/gain pathways unmade. As explained in Section 8.2.3 the NER measurements made in this study include respiration losses from below-ground crop and tree roots, and therefore for the true impact on the total carbon balance to be assessed, measurement of the PP of below-ground root biomass is also required.

This study has assessed the impacts of converting from traditional arable crops to bioenergy plantations during the growth phase of the bioenergy plantation; however it has not assessed the implications on the total carbon balance following crop harvest. Although some of the carbon sequestered into bioenergy crops will be released into the atmosphere upon energy generation, this is likely to replace emissions which would otherwise occur from the burning of fossil fuels. The carbon savings from the cessation of fossil fuel burning would therefore need to be included in any analysis to clarify the impact on the total land carbon balance. Bioenergy crops are also increasingly being targeted as a potential fuel source for biochar (Chapter 7), and the longevity and stability of biochar in soils as well as its impacts on crop productivity and carbon sequestration into plant biomass will need to be assessed. Although complete assessment of these factors is beyond the scope of this chapter, the results from Chapter 7 do indicate that biochar has the potential to be a large carbon sink, and once clarified the results observed in this study can provide a baseline onto which these values can be added to or subtracted from in order to determine the complete carbon savings possible with bioenergy plantation and the replacement of arable cropland.

8.5 Conclusions

Carbon losses to the atmosphere from the soil and surface vegetation in this trial have been shown to be greater from SRC willow plantations than from arable land. These losses were found to increase with SRC willow plantation age, with an annual estimated loss of 3.77 t C/ha from a plantation planted in 2006, an annual loss of 2.59 t C/ha from a plantation established in 2008, and an annual loss of 2.55 t C/ha from an arable field. Measurement of the %SOC of soil samples taken from beneath the older SRC willow plantation correlates with the NER measurements, with the lowest %SOCs found under this plantation. These results suggest that if the major aim of land-use change is to preserve or increase SOC stocks then conversion of arable land to SRC willow plantation should not be made. If however land-use change is to be made to increase total land-carbon sequestration including that in above and below-ground biomass then the impacts of arable land conversion to SRC willow plantation are more uncertain, and require more research. A combination of measurements made in this study and above-ground carbon sequestration values from the literature suggest that conversion of arable land to SRC willow will increase the size of the land carbon sink. A lack of measurements relating to the PP of below-ground root biomass however means that no firm conclusions can be made regarding the impact of SRC plantation on the total carbon balance until much more research is undertaken. For the full impact on atmospheric carbon to be established the end use of the products of the alternative land-uses, the carbon emissions that could be offset through these end uses, and the potential by-products which could be produced and used must also be firmly established.

Chapter 9

Biochar application to pasture

9.1 Biochar and grasslands: an introduction

The majority of research into biochar, although still in its infancy, is associated with incorporation of the biochar material into mineral soils that are low in SOC and in arable land-use. In addition some research has also considered the impact of biochar incorporation into more organic-rich soils (Wardle et al., 2008; Chapter 7) used for forestry plantation. Although there are still many uncertainties and issues to be resolved before biochar can be used as a carbon sequestration tool on a world-wide basis in these land-uses, there are many positive outcomes from this research, revealing a large potential to incorporate biochar into UK arable and forestry land. No published research has however assessed the impacts of application to grassland, which constitutes approximately 70% of UK agricultural land (Figure 9.1) and 59.27 % of the NT's Wallington Estate (Figure 1.2).

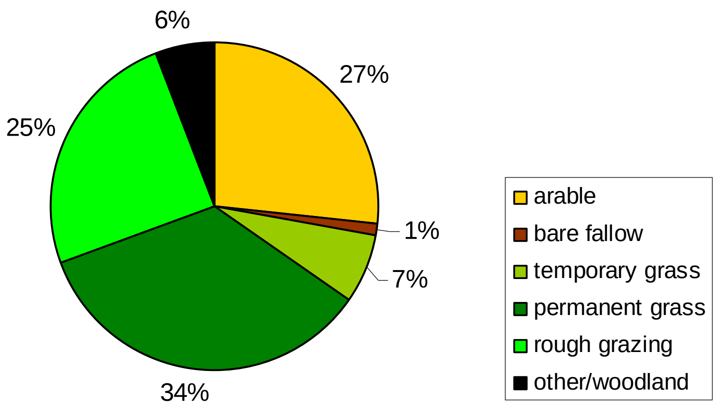

Figure 9.1 Grassland contribution to UK agricultural land-use (adapted from Angus et al., 2009)

If biochar application is limited to soils in arable and forestry land-use then the area of land to which it can be applied is severely limited (Figure 1.2). Not only is the area of arable land in the UK and on the Wallington Estate currently small in relation to total agricultural land area, there is also the potential for this area to decrease further in the future if land-use change to avoid carbon emissions, or to increase SOC stock (see Chapter 6) is made. To utilise biochar's huge potential would therefore require application to rough and improved grasslands used for grazing livestock.

Although the results of Chapter 7 suggest that biochar incorporation into soil will result in the land into which it is incorporated becoming a carbon sink, the same can not be assumed to occur when biochar is applied to grasslands until trials on such a land-use have been undertaken. The carbon sink resulting from incorporation into arable and forestry land in Chapter 7 was the consequence of burying large amounts of carbon within the soil, without causing losses of gaseous carbon, or losses of DOC of the same extent or magnitude as the carbon being sequestered in charcoal. Along with the carbon sequestration benefits of biochar incorporation in such land-use and soil types, the results of Chapter 7 also reveal other environmental co-benefits with charcoal incorporation, including reduced nitrate leaching from the soils, and greater soil pH. Although no improvement in crop productivity was observed with biochar incorporation in Chapter 7, other studies in the literature have found such results (e.g. Lehmann et al., 2006). Despite these positive outcomes there are a large number of uncertainties and potential consequences as discussed in Chapter 7, and these uncertainties and consequences will need to be dismissed before the uptake of biochar can be encouraged on a universal scale.

If biochar is to be applied to grasslands as a top-dressing, it is essential to ensure that the potential consequences relating to its incorporation into soils are not also potential consequences when biochar is applied directly to grassland surfaces. In addition, it is necessary to determine whether the positive aspects and potential carbon sequestration achievable through biochar incorporation in arable soils is true also of biochar spread directly onto pasture. Assessing the impact on the carbon balance and water chemistry following lump-wood charcoal (used in this study as a substitute for biochar) application to grazed pasture is the focus of this chapter. As there is currently no official 'biochar' product, the properties of 'biochar' can vary greatly depending on the source material and the method and temperature of production. Throughout this chapter it should therefore be realised that the results gained from this research should not be assumed to be representative of all types of 'biochar', with the production temperature and properties of the biomass heated having a large impact on the resulting biochar properties.

9.1.1 Biochar: a review of the literature

All published research relating to biochar and soils is associated with incorporation of the material by mixing into the topsoil. A literature review regarding the impact of biochar incorporation into soils in arable and forestry land-use is presented in Chapter 7, along with the results from a trial undertaken as part of this thesis. Although Verheijen et al (2010) state that the top-dressing of biochar is being considered as a method of application, in-depth literature searches were unable to find any such published studies. Verheijen et al. (2010) also state that the rates of natural incorporation into the topsoil following surface spreading methods are unknown, and although there is a lack of information relating to both biochar application to grasslands, and the more specific aspect of surface spreading, there is evidence of charcoal being applied to grasslands several decades ago. In a magazine article by the US Golf Association written in 1943 (Association Golf, U.S, 1943) charcoal is described as a “must” for maintaining putting green surfaces, and is reported to be responsible for producing a thick and healthy looking turf. This improvement in productivity was attributed by the authors to a deeper rooting system, increased water efficiency and improved drainage. It must be noted however, that this charcoal was not simply spread onto the grass surface, but was injected into the ground beneath. No studies relating to the top dressing of biochar with zero incorporation have been identified.

9.1.2 Chapter aims

The aim of this chapter was to establish the degree of carbon sequestration achievable when lump-wood charcoal (as a substitute for biochar) is applied as a top-dressing to the surface of pasture land in north east England. In order for biochar application to this land-use to be considered a successful tool for mitigating climate change, it will need to be a greater sink than source of carbon, and so the size of the carbon sink with application needs to be established. The amount of carbon applied to grassland in biochar could be considered to represent the size of the carbon sink, but it is also necessary to establish any additional carbon sequestration that could occur as a result of increased PP, and changes in above-ground biomass. This value then needs to be combined with that of the carbon in biochar to produce a value for the total carbon sequestration achievable through this land-management change. The size of carbon emission pathways in the form of NER and DOC loss also need to be established, to ensure that the application of biochar does not enhance these fluxes to an extent that the carbon sink achieved through application and potential increased PP is not counteracted or superseded. In addition to the measurement of carbon fluxes, it was also aimed to measure any change in %SOC resulting from biochar application, although as described in Section 4.2.3 it was realised that a large spatial variability in %SOC may result in any change not being apparent and detectable over such a short time period.

In compliance with the aims of the NT set out in Section 1.1.2, it was a further objective to assess the impact of charcoal application on soil and run-off water chemistry. This was done to identify any retention or loss of fertiliser with charcoal application, and help determine if the use of artificial fertilisers can be reduced to provide environmental and economic benefits, or whether the application of biochar to pasture could be detrimental to the environment. It was also hoped to establish the impact of charcoal application on soil pH, as the results from previous research (see Chapter 7) suggest increases in soil pH upon charcoal incorporation and a potential reduction in the need for lime application. As with all other measurements to be made, the impacts on soil pH under grassland may not correspond with results from soil incorporation, making this research specific to surface spreading of vital importance.

To achieve the objectives of this chapter a plot trial where different amounts of lumpwood charcoal were applied to grazed pasture was undertaken in north east England, with an aim to measure the following:

- pre and post trial %SOC, to identify any change following one year of treatment
- gaseous fluxes of carbon in the form of NER, NEE and PP every fortnight over an annual period
- a change in above-ground biomass accumulation following one year of treatment
- pre and post trial soil pH to identify any change following one year of treatment
- soil water pH, electrical conductivity, anion and DOC concentrations every fortnight, to identify carbon losses and nutrient leaching from the soil
- run-off water pH, electrical conductivity, anion and DOC concentrations on a fortnightly basis, to identify carbon losses and nutrient run-off from the grassland surface

Alongside these measurements, for the purpose of inclusion as covariates in any analysis, it was also aimed to measure water table depth, surface-air temperature and PAR.

9.2 Materials and Methods

9.2.1 Study site

The trial was undertaken on a field on the NT's Wallington Estate in north east England (Section 1.2). The field was chosen due to its uniform land-use, topography and limited variation in soil series and altitude, allowing a largely controlled experiment to be undertaken with few uncontrolled variables (Figure 9.2).

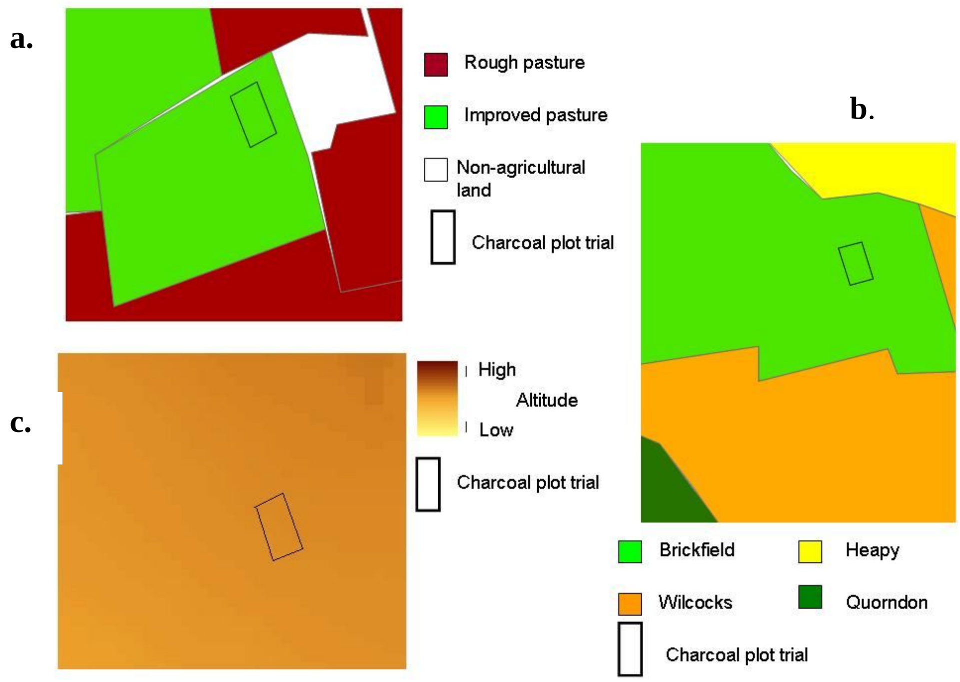

Figure 9.2 a. Plot location under a constant land-use; b. plot location under a constant soil series; c. Plot location with little variation in altitude. Where RP = rough pasture; IP = improved pasture; Heapey, Wilcocks, Brickfield and Quorndon refer to soil series

Other than in winter, and in May and June when the grass was being grown for silage, the field was grazed continuously by sheep. Grazing livestock were present throughout the trial period as it was undertaken on a working farm and grazing was considered indicative of a real-world situation. The fact that grazing is an uncontrolled variable in this trial means it must be taken into consideration in the analysis of all results, as it is possible, although not known, that sheep may have favoured and grazed certain plots more than others.

9.2.2 Trial design and set-up

Charcoal treatments were applied to a total of eight plots, each measuring 5m x 5m on 23rd July 2009. The plots were marked out using pegs and string, and separated from each other by 2m strips to which no charcoal was applied. These buffer strips were created to minimise any contamination from nearby plots which could result during charcoal application, and as a result of wind and rain dispersing the charcoal during the trial period. To ensure that all charcoal treatments were subjected to the same conditions the plots were laid out in a uniform area of the field, with very little or no variation in slope angle, vegetation type or soil type. The eight plot design consisted of 3 levels of charcoal, each in replicate, and two plots to which no charcoal was applied, to act as a control. The plots were laid out in a completely randomised design to limit any bias which may have resulted from field position.

The varying amounts of charcoal applied to grassland in this trial were:

1. 0 kg/plot (control)
2. 20 kg/plot (10,000 kg/ha)
3. 80 kg/plot (32,000 kg/ha)
4. 160 kg/plot (64,000 kg/ha)

The charcoal used was charcoal fines supplied from a local wood-smith, and was produced from burning hardwood deciduous tree species in a purpose made steel ring kiln at 400 °C, with restricted oxygen intake for 12 hours. The amount of charcoal to apply was chosen based on the reasoning described in Section 7.2.2 and the results of Chapter 7, but considering also the fact that no trials have been previously undertaken where application has involved the top dressing of biochar without incorporation. As in Chapter 7 only a low rate was applied as the minimum amount, to allow assessment of the effects of applying the level of carbon needed to counteract the Estates current carbon emissions. If spread over an area of pasture totalling 3 km^2, and assuming no biochar decomposition or native SOC loss, this could make the Wallington Estate carbon neutral for 2 years. Assessment of the effects of application at this low dosage was deemed vital, as annual application of higher amounts may not be feasible depending on the economic costs of biochar application (assessment of which is beyond the scope of this chapter). It was hence considered necessary to establish whether these low application rates could provide immediate positive effects regarding grass productivity to encourage farmers and landowners in its application, if only applied at this low rate on an annual basis. The maximum charcoal treatment incorporated into soil in Chapter 7 showed no significant increase in NER other than in the first week following application. This indicated that incorporation of biochar at its lowest application rate could be undertaken annually for a total of 26 years, or alternatively as a one-off application making the Wallington Estate carbon neutral for this length of time. Incorporation into arable soils at this rate was found to have no negative impacts on crops or water quality, and in addition provided a significant reduction in nitrate leaching (see Section 7.3.4). Investigation of the effects of application of a similar rate directly onto the surface of pasture was deemed necessary as it is unknown whether charcoal applied to grasslands will cause soil to behave in a similar manner. Surface spreading of charcoal onto pasture in such large amounts was however considered inappropriate due to the extent to which the charcoal would restrict PAR from reaching the grass surface, and could prevent grass growth and have severe negative impacts if not immediately incorporated into the soil profile by natural processes. For this reason a lower maximum charcoal application rate of 64,000 Kg/ha was chosen, as although it reduced the amount of PAR reaching some of the pasture surface (Figure 9.3) large amounts of grass were still able to penetrate through the applied charcoal, and it was thought that the surplus charcoal would be incorporated into the soil profile by natural processes. It should also be noted that application rates to arable and forest soils chosen in Chapter 7 were based on the assumption that biochar would only be applied to the current area of arable land (~ 3 Km^2) on the Estate. With the same objective to counteract the Estates current carbon emissions (794.54 t C/yr) in this trial, with biochar application to grassland, a much larger area of improved and rough pasture could be utilised. As such, the maximum charcoal application rate applied in this trial to make the Estate carbon neutral, although smaller than that in Chapter 7, will in-fact make the Estate carbon neutral for a much longer length of time. With pasture covering a total area of 32.6 Km^2 (Section 1.2.2), and assuming a charcoal carbon concentration of 80% (Lin and Hwang 2009), this will produce a carbon sink of 166,912 t, thus application at this rate will make the Estate carbon neutral for 210 years (assuming no charcoal carbon degradation and no loss of native SOM).

Charcoal was applied directly to the pasture surface following a silage cut, spread using a rake, and equally distributed across the plot. Photos of the plots immediately following charcoal application are presented in Figure 9.3 and Figure 9.4.

Figure 9.3 Trial appearance immediately following charcoal application

Figure 9.4 Appearance of plots treated with 64,000 Kg/ha and 32,000 Kg/ha immediately following charcoal application

To allow measurement of NER, NEE and PP as described in Section 9.2.4, three soil respiration collars were installed into each plot, with each collar constructed from a six inch diameter drain pipe of 10cm length. The respiration collars were inserted in random locations within each plot to a depth of approximately 5 cm, leaving 5 cm protruding from the surface. These collars were installed on the day of charcoal spreading, and were left in place for two weeks before any measurements were taken, for the reasons described in Section 4.2.2.

9.2.3 Pre and post trial %SOC

Although it was thought unlikely that a difference in %SOC would be evident between plots treated with different amounts of charcoal after only one year (see Section

4.2.3) it was still considered important to investigate whether any differences were apparent. The large spatial variability and natural variation in %SOC often found beneath small areas of land meant that in addition to comparing %SOC from the control and charcoal treated plots at the end of the trial, it was also decided to measure %SOC beneath each plot prior to charcoal application, and to compare these measurements to the %SOC beneath each plot at the end of the trial. The large spatial variation often found in %SOC (Saby et al., 2008; Tolbert et al., 2002; Chapter 2) meant that a high sampling density was required, to obtain an accurate mean %SOC for each plot. A total of six soil samples were therefore taken from a depth of 20 cm (for reasons see Section 4.2.3) beneath each plot in June 2009, prior to charcoal application. These samples were taken from six random locations beneath each plot, using a soil auger and collecting a sample from a depth of 18-22 cm. The same procedures and sampling density were used to collect the samples one year after charcoal application. Following soil collection all samples were transported immediately back to the laboratory where they were dried overnight at 105°C, and treated and analysed for %SOC following the methodology described in Section 2.2.2.1. This resulted in the total analysis of 96 soil samples (48 pre trial and 48 post trial), with 12 from beneath each charcoal treatment on both occasions, allowing statistical analysis of the difference in %SOC between treatments to be undertaken.

9.2.4 Carbon flux measurements

9.2.4.1 NER

Measurements of NER were made from the ground surface every fortnight from August 2009 to July 2010, and include respiration from the above and below-ground vegetation, roots and soil. These measurements represent the total amount of carbon released from the ground surface in g C/m^2/hr. The first measurements were made on 6th August 2009 between 1000 and 1600 hours, and every fortnight following, to gain an accurate estimate of the seasonal variation in flux over an annual period. As this measurement represents the flux of carbon from the land to the atmosphere, it is always a positive number, the more positive the greater the release of carbon to the atmosphere.

NER was measured using a portable IRGA (PP systems EGM-4, Hitchin, UK) following the methodology described in Section 4.2.4.1.

9.2.4.2 NEE

Measurements of NEE were also made from the ground surface on a fortnightly basis from August 2009 to July 2010. These represent the difference between the total amount of carbon released from the ground surface and that taken up from the atmosphere in PP. Measurements began on 16th August 2009 and were made between 1000 and 1600 hours. Measurements were made following the methodology outlined in Section 4.2.4.2. NEE measurements refer to g C/m^2/hr and were either a negative or positive value depending on whether the combination of soil/vegetation was a sink or source of carbon.

9.2.4.3 PP

As with NER and NEE, calculations of PP were made every fortnight from August 2009 to July 2010. These calculations represent the total amount of carbon taken in from the atmosphere by vegetation growing on the soil surface, and the below-ground vegetation and roots. PP is represented as a negative number, the more negative the number the greater the uptake of carbon from the atmosphere. For information on the calculation of PP refer to Section 4.2.4.3.

9.2.4.4 Surface-air temperature and PAR

Measurement of surface-air temperature at the time of each IRGA reading was required, as indicated in Section 4.2.4.1. This was recorded along with every IRGA reading as described in Section 4.2.4.4. As a variable over which there could be no experimental control it was also important to measure and record PAR, as this can vary greatly over very short timescales due to variations in weather conditions and cloud cover. Measurement of PAR was undertaken along with every CO_2 reading by a PAR sensor located within the chamber.

9.2.5 Biomass carbon stocks

Although fortnightly measurements of PP using IRGAs will reveal the mean PP under different charcoal treatments for each period of measurement, it was also thought beneficial to sample the total above-ground biomass accumulation from each plot one year after charcoal application. These measurements would show how total accumulation varied with the amount of charcoal applied, and can be used to support the IRGA readings. Total above-ground biomass from each charcoal treatment was measured at the end of the trial in July 2010, following the methodology described in Section 4.2.5.

9.2.6 Soil water chemistry and water table depth

On the day of charcoal application a 100 cm long dipwell was inserted at least 90 cm into the ground of the centre of each plot as described in Section 4.2.6, and water table depth measured on every visit to the site (also described in Section 4.26). In addition to water table depth measurement, a sample of groundwater was extracted from the dipwells for water chemistry analysis on each visit. Each sample was analysed for pH, electrical conductivity, DOC concentration, chloride concentration, nitrate concentration and phosphate concentration following the methodologies described in Section 4.2.6.

9.2.7 Run-off water chemistry

Following charcoal application three run-off traps were installed into each plot to collect water draining from the ground surface, and to assess any effects of charcoal application on run-off water DOC concentration, pH, nitrate concentration, phosphate concentration and chloride concentration. The run-off traps were constructed and installed as described in Section 4.2.7.

9.2.8 Pre and post trial soil pH

Soil pH was measured on every soil sample collected and analysed for %SOC in Section 9.2.3. This resulted in 96 soil pH measurements, 48 pre trial and 48 post trial, consisting of 12 samples from each of the charcoal treatments on both occasions. As with the samples for %SOC analysis, soil pH from beneath the control plots at the end of the trial was compared to soil pH from the plots treated with charcoal at the end of the trial. The same issues regarding large spatial variations in %SOC were however also likely to apply to the natural variation in soil pH. It was for this reason that soil pH measurement was undertaken on all soil samples prior to charcoal application, so that in addition to comparing post trial pH from plots treated with charcoal to that of the control plots, a comparison of soil pH pre and post charcoal application could also be made if soil pH varied significantly across the field site prior to treatment.

9.2.9 Statistical analysis

All analyses of %SOC, NER, NEE, PP, soil pH and soil water chemistry were performed using Minitab 14 statistical analysis software. Comparison of %SOC between plots to be treated with varying amounts of charcoal prior to the start of the trial was done by one-way ANOVA, as the nature of the study site meant that all other variables were controlled (see Section 9.2.1). The same statistical analysis was used to compare %SOC between treatments at the end of the trial, and to compare the difference in %SOC pre and post trial. The r^2 values generated by ANOVA represent the between *group sum of squares* divided by the *total sum of squares,* with a large r^2 value thus indicating that a large fraction of the variation in the independent variable can be explained by the categorical variable/treatment. The r^2 value represents the proportion of the total variation explained by the difference in the means. Analysis of soil pH between charcoal treatments was undertaken in the same way to that of %SOC. Analysis of NER, NEE and PP between land-uses was undertaken using ANOVA, with ANCOVA then used to establish if these land-use controls remained significant with uncontrolled experimental variables held constant. The factors investigated in this study as controls on NER, NEE and PP were charcoal treatment and month of measurement. A variation in surface-air temperature with measurement meant that this was entered as a covariate in all analyses. Water table depth and PAR were also considered as covariates in all analyses, and PP was considered as a covariate in the analysis of NER. Main effects plots were generated to display the adjusted means of the variable under study for each of the independent factors. The main effects plots show the mean for each factor with the effects of other variables removed. Each point on the main effects plots is the mean of all measurements taken over the trial period for that factor, with the horizontal line representing the overall mean of the entire dataset. Any significant differences in NER identified with charcoal treatment then led to MLR analysis to establish the respective controls on NER for each respective treatment. Analysis of DOC in both soil water and run-off water, and its variation with charcoal treatment, was undertaken using ANOVA, with charcoal treatment and month of year entered as factors in the analysis. ANCOVA was then undertaken to establish whether charcoal treatment remained significant with other variables held constant. In this analysis water table depth, soil/run-off water pH, soil/run-off water electrical conductivity and the concentration of each of the analysed soil/run-off water anions were included as covariates. Analysis of the variation in nitrate concentration with charcoal treatment in both soil and run-off water was undertaken using a GLM and ANOVA, with charcoal treatment and month of measurement entered as factors in the analysis. ANCOVA was then undertaken on these samples, with water table depth and run-off/soil water pH included as covariates. Before any analysis was undertaken it was ensured that all data were normally distributed. Any significant difference identified by ANCOVA or ANOVA was then post-hoc tested using the Tukey test, to identify between which factors significant differences ($p < 0.05$) in %SOC, NER, NEE, PP, soil and run-off water DOC, soil and run-off water nitrate concentration and soil and run-off water pH and soil pH occurred. The effect of charcoal treatment on soil and run-off water phosphate concentration could not be measured due to insufficient concentrations to allow analysis.

9.3 Results

All raw data relating to %SOC, CO_2 flux measurements and soil and run-off water chemistry can be found in Appendix 9.

9.3.1 Change in %SOC with charcoal application

Collection of soils prior to charcoal application resulted in a total of 46 samples for %SOC analysis. This consisted of 12 from the control plots, 10 from the 10,000 Kg charcoal/ha treatment, 12 from the 32,000 Kg charcoal/ha treatment, and 12 from the 64,000 Kg charcoal/ha treatment. Mean initial %SOC varied across the treatments from a low of 3.33% to a high of 3.46%. The lowest mean %SOC of 3.33% was found under the plots to be treated with 32,000 Kg charcoal/ha, and the highest of 3.46 % under the control plots. One-way ANOVA revealed that there was no statistically significant difference in %SOC between any of the treatments, shown in Figure 9.5.

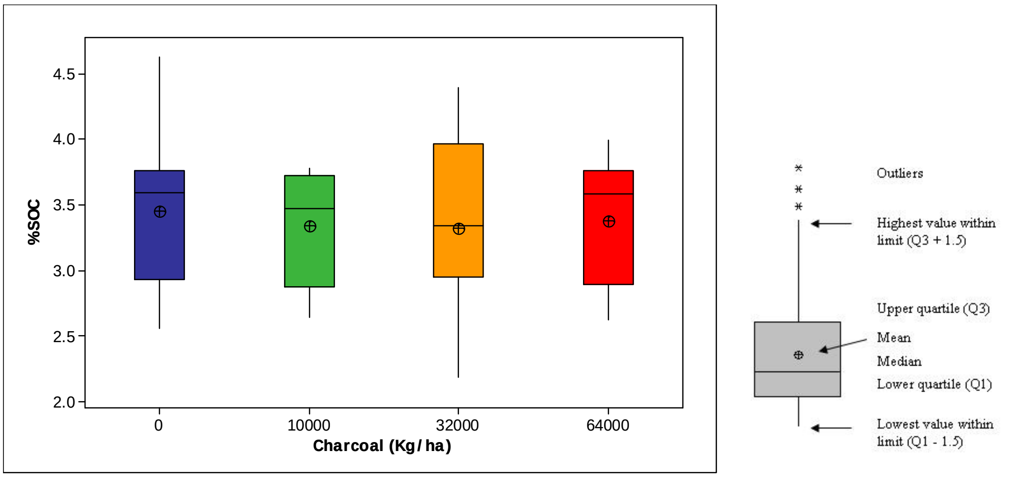

Figure 9.5 The variation in %SOC under plots prior to charcoal application

Collection of soils one year after charcoal application resulted in 27 samples for analysis of %SOC. This consisted of 9 from the 10,000 Kg charcoal/ha treatment, and 6 from all other treatments, with %SOC varying from a low of 2.26% to a high of 3.46%. The lowest mean %SOC of 2.26% was found under the control plots and the highest mean %SOC of 3.46 % under the 32,000 Kg charcoal/ha treatment (Figure 9.6).

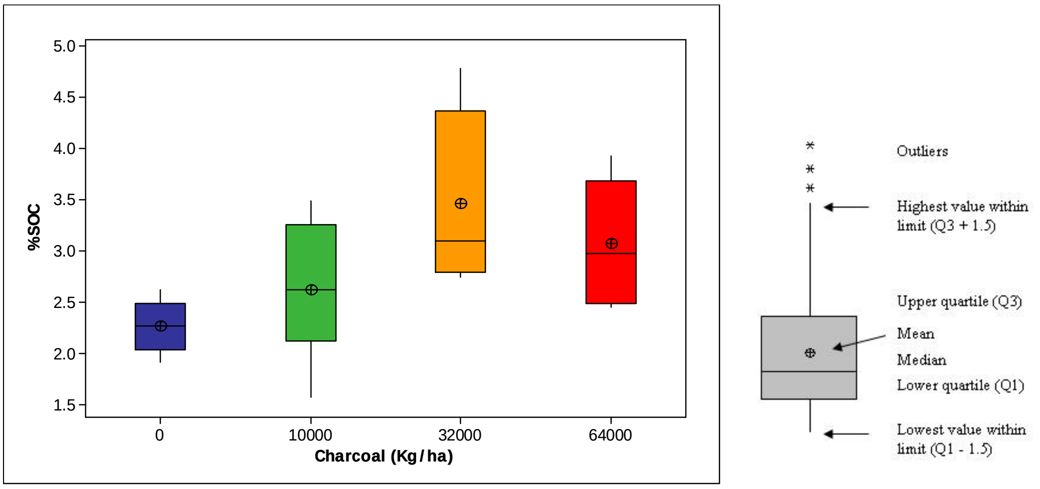

Figure 9.6 The variation in post trial %SOC in and between charcoal treatments

One-way ANOVA revealed a statistically significant difference in post-trial %SOC between charcoal treatments, and post-hoc analysis was undertaken to establish between which treatments this significant difference did occur. These results indicated that the only significant difference was that between the control plots and the 32,000 Kg charcoal/ha treatment, with the control plots having a significantly lower %SOC.

Although %SOC did not vary across the field site prior to charcoal application it was decided that comparison of before and after treatment %SOC should still be made. This revealed a significantly lower %SOC under the control plots one year after initial sample collection, a significantly lower %SOC under the 10,000 Kg charcoal/ha treatment one year after charcoal application, but no significant difference between the post and pre-trial %SOC under the 32,000 Kg charcoal/ha treatment or 64,000 Kg charcoal/ha treatment. These changes in %SOC over the one year trial period can be observed in Figure 9.7.

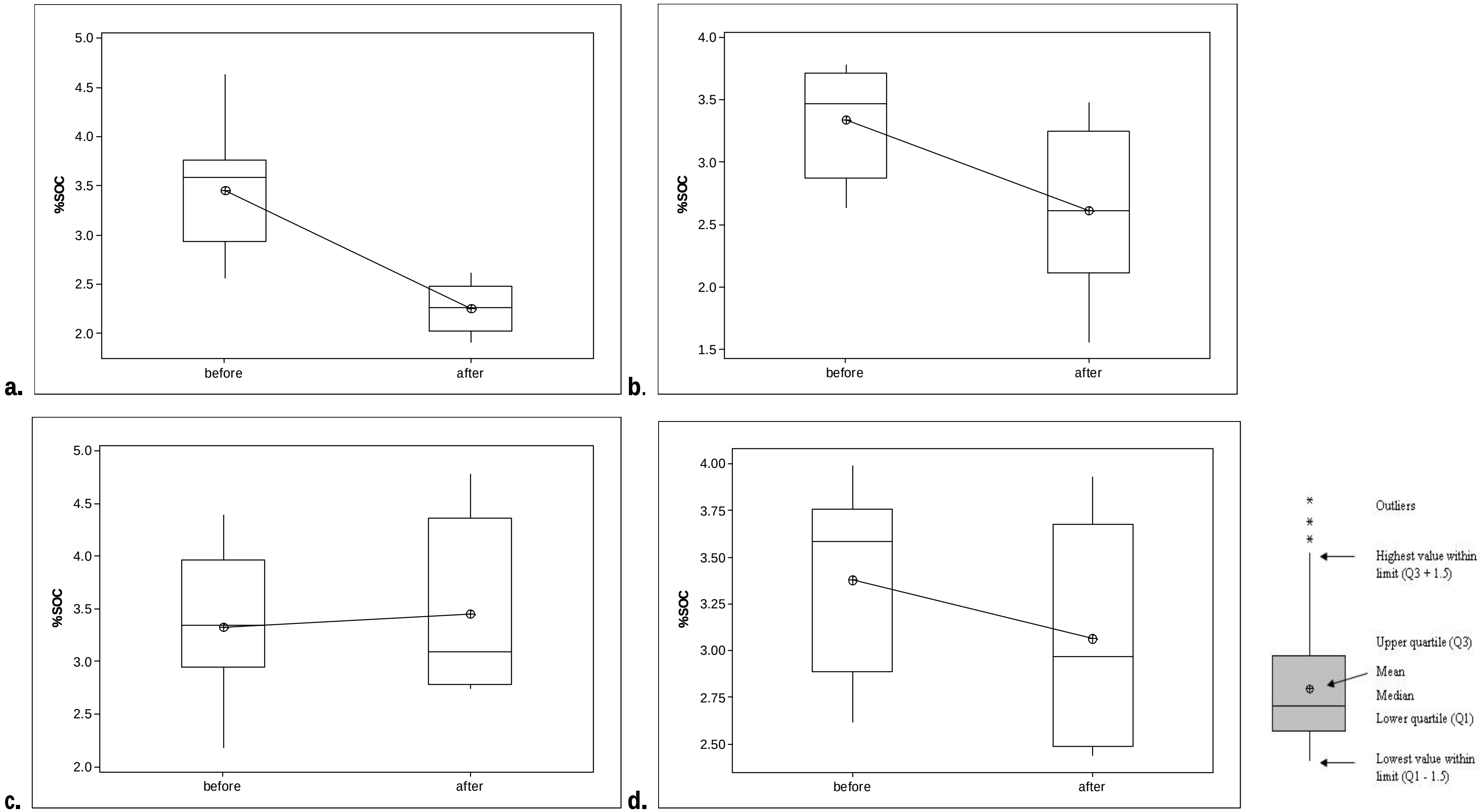

Figure 9.7 A change in %SOC following one year of charcoal application under plots treated with a. zero charcoal; b. 10,000 Kg charcoal/ha; c. 32,000 Kg charcoal/ha; d. 64,000 Kg charcoal/ha

9.3.2 *Gaseous carbon fluxes*

9.3.2.1 NER

Fortnightly monitoring of NER over the one year period resulted in 462 NER flux measurements. This consisted of 116 measurements from the control plots, 120 from the 10,000 Kg charcoal/ha treatment, 112 from the 32,000 Kg charcoal/ha treatment, and 114 from the 64,000 Kg charcoal/ha treatment. The slight inconsistency in the number of measurements from each treatment was a result of the inability to locate a number of the respiration collars following dense grass growth when the field was being grown for silage.

Although NER was not normally distributed its distribution did not improve following log transformation, therefore NER was not log-transformed in any analysis. For the same reasons none of the other covariates used in the analysis of NER were log-transformed. Two-way ANOVA revealed that both charcoal treatment and week of measurement had a significant impact, and could explain some of the variation in NER. The interaction between charcoal treatment and measurement week was also found to be significant, and inclusion of this interaction in a GLM revealed that 83.76% of the variation in NER could be explained. The significant effect of the interaction between measurement week and charcoal treatment revealed that the impact of treatment was not constant over the entire measurement period, and that the treatment from which the greatest NER flux occurred varied with time of year. The significant effect of charcoal treatment and its variation over the trial period is demonstrated in Figure 9.8.

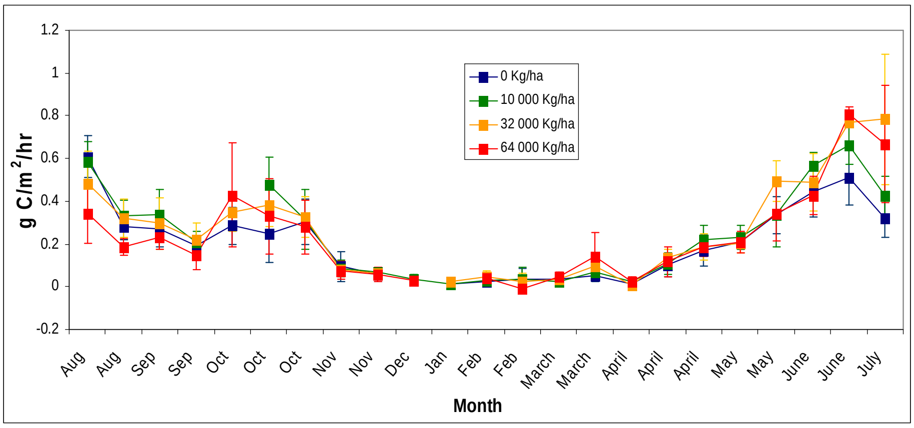

Figure 9.8 The variation in NER with charcoal treatment and measurement week, where error bars represent the standard deviation from the mean

The covariates surface-air temperature and water table depth could not explain any of the additional variation in NER, however PP in combination with charcoal treatment, measurement week, and the charcoal treatment/measurement week interaction could explain 89.39% of the variation in NER. With the covariate PP included, the factor charcoal treatment continued to have a significant effect. Post-hoc analysis of the results indicated that over the entire measurement period there was a significantly greater NER flux from the 32,000 Kg charcoal/ha treatment and the 64,000 Kg charcoal/ha treatment than from the control plots, but no significant difference between any other treatments. The main effects plot, revealing the difference in mean flux with charcoal treatment when other factors are controlled, is displayed in Figure 9.9. A mean flux of 0.23 g C/m^2/hr from the control plots can be compared to a mean of 0.27 g C/m^2/hr from the 32,000 Kg charcoal/ha treatment.

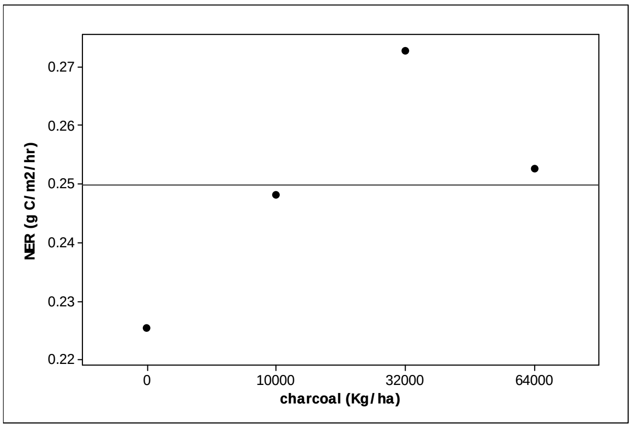

Figure 9.9 Mean trial NER from different charcoal treatments

Further post-hoc analysis was undertaken to establish in which weeks the statistically significant difference in NER flux occurred. This revealed a significantly lower NER from the control plots than from the 10,000 Kg charcoal/ha treatment in the sixth measurement week (October), and a significantly lower flux from the control treatment than from the 32,000 Kg charcoal/ha treatment and 64,000 Kg charcoal/ha treatment respectively in the final measurement week (July).

With a significant difference in NER established between charcoal treatments, the complete dataset was split into individual treatments, and MLR analysis undertaken on each to identify the respective controls on NER with treatment. Under all treatments including the control, variation in NER could be explained by a variation in surface-air temperature, PP and water table depth. The size of the effect of each variable on NER does however differ with treatment, indicated by the coefficients displayed in Table 9.1.

Charcoal treatment (Kg/ha)	Surface-air temperature	PP	Water table depth	r^2	n
0	0.011	-0.303	0.001	72.9	113
10,000	0.008	-0.228	0.003	77.3	118
32,000	0.075	-0.378	0.003	74.1	111
64,000	0.009	-0.540	0.002	69.2	111

Table 9.1 MLR equation co-efficients, indicating the effect of different variables on NER under different charcoal treatments, where: r^2: represents the amount of variation in NER that can be explained by the independent variables; n: represents the number of observations included in the regression

9.3.2.2 NEE

Fortnightly monitoring of NEE over the one year period resulted in a total of 458 NEE measurements. This consisted of 116 from the control plots, 118 from the 10,000 Kg charcoal/ha treatment, 112 from the 32,000 Kg charcoal/ha treatment and 112 from the 64,000 Kg charcoal/ha treatment, with a slight inconsistency in the number of measurements from each treatment being the result of the reasons discussed in Section 9.3.2.1.

As in the analysis of NER, none of the covariates were log transformed, however an extra covariate, photosynthetically active radiation (PAR), was included in this analysis. The distribution of PAR became more normal upon log transformation; therefore this log transformed value was used in all analysis of NEE. Statistical analysis via two-way ANOVA revealed that variation in both charcoal treatment and measurement week could explain some of the variation in NEE. The interaction between measurement week and charcoal treatment was also significant, and these factors in combination could explain 55.27% of the variation. Although surface-air temperature was returned as a significant control when included as a covariate, this significant effect became insignificant when PAR was also included, suggesting that PAR and temperature may be collinear. The inclusion of water table depth in the analysis could not explain any of the further variation in NEE. In combination, the factors charcoal treatment, measurement week, the charcoal treatment/measurement week interaction and PAR could explain 57.48% of the variation. With the covariate PAR included in the analysis the significant effect of charcoal treatment remained. Post-hoc analysis of the data revealed a significantly lower (less negative) mean NEE over the trial period from the 64,000 Kg charcoal/ha treatment than from the control plots, and a significantly lower NEE from the 64,000 Kg charcoal/ha treatment and 32,000 Kg charcoal/ha treatment than those plots treated with 10,000 Kg charcoal/ha. The main effects plot, showing the difference in mean flux over the measurement period when other variables are controlled, can be seen in Figure 9.10.A mean NEE of -0.04 g C/m^2/hr occurred beneath the 10,000 Kg charcoal/ha treatment compared to -0.01 g C/m^2/hr beneath the 64,000 Kg charcoal/ha treatment.

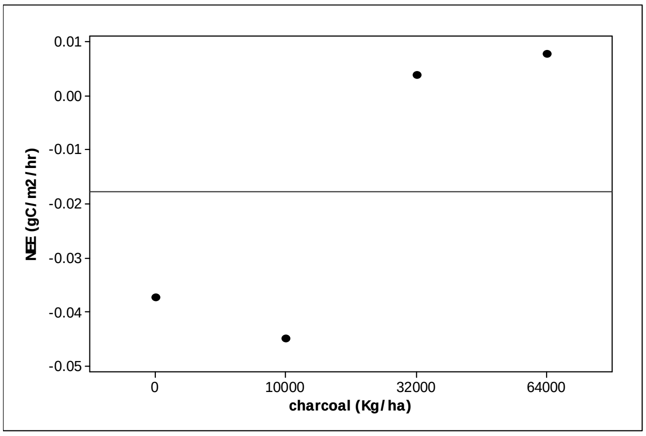

Figure 9.10 Main effects plot showing the mean trial NEE from different charcoal treatments

Further post-hoc analysis to establish under which weeks the significant difference in NEE between treatments occurred revealed a significantly greater NEE under the control plots than the 64,000 Kg charcoal/ha treatment in week one, and a significantly greater NEE under the 10,000 Kg charcoal/ha treatment in the final week than under the 32,000 Kg charcoal/ha treatment. The variation in NEE under the different charcoal treatments and this variation with measurement week can be observed in Figure 9.11.

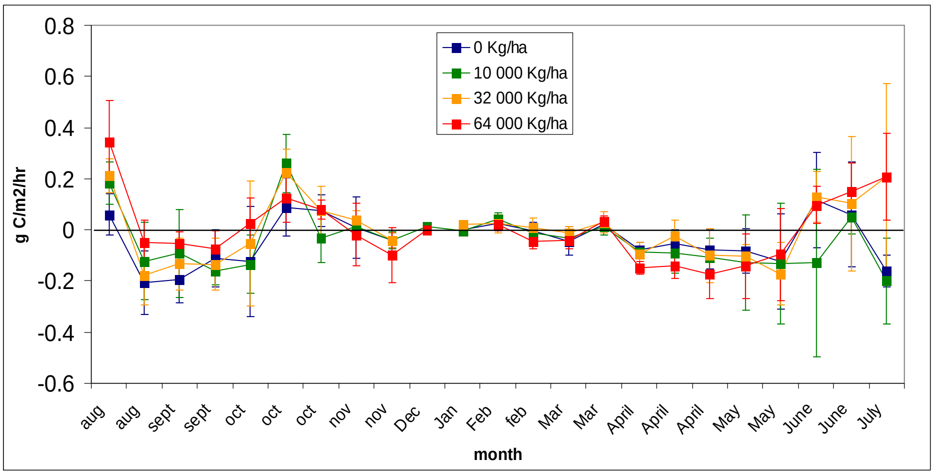

Figure 9.11 The variation in NEE with charcoal treatment and measurement week, where error bars represent the standard deviation from the mean

9.3.2.2 PP

Calculation of PP using the NEE and NER measurements made in Section 9.3.2.1 and Section 9.3.2.2 resulted in a total of 457 PP values. This consisted of 114 from the control plots, 119 from the 10,000 Kg charcoal/ha treatment, 112 from the 32,000 Kg charcoal/ha treatment, and 112 from the 64,000 Kg charcoal/ha treatment.

Statistical analysis via two-way ANOVA revealed that variation in charcoal treatment and measurement week could together explain 58.68% of the variation in PP, and that they were both significant controls. The interaction between charcoal treatment and measurement week was also found to be significant, and inclusion of this interaction in the analysis revealed that 69.50% of the variation in PP could be explained. Inclusion of the covariates surface-air temperature and PAR revealed that variation in both could explain some of the further variation in PP, and that combined with charcoal treatment, measurement week and the charcoal treatment/measurement week interaction could explain 70.42% of the variation in PP. With inclusion of these covariates the significant effect of charcoal treatment remained. Post-hoc analysis of the data revealed that the only significant difference in PP between treatments was that of greater PP under the 10,000 Kg charcoal/ha treatment relative to the 64,000 Kg charcoal/ha treatment. This difference in PP can be observed in Figure 9.12, where the main effects plot indicates a mean PP of -0.32 g C/m^2/hr below the 10,000 Kg charcoal/ha treatment compared to a mean of -0.25 g C/m^2/hr from the 64,000 Kg charcoal/ha treatment when other variables are controlled.

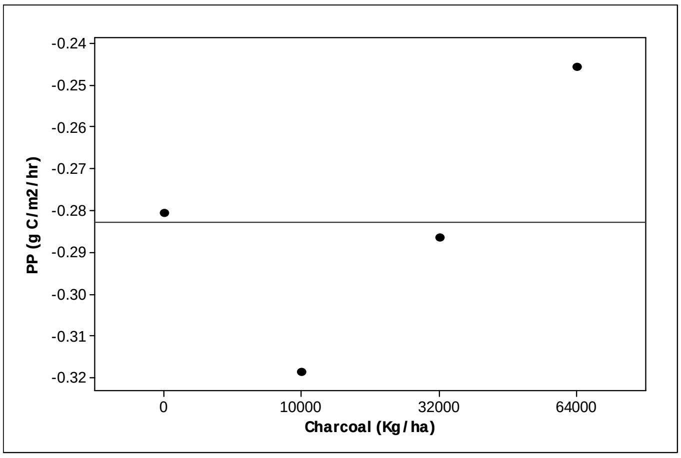

Figure 9.12 Main effects plot showing mean trial PP under different charcoal treatments

Further post-hoc analysis of the data was then undertaken to identify in which weeks the significant difference in PP between charcoal treatments had occurred. This revealed significantly greater PP under the control plots in the first measurement week than under the 64,000 Kg charcoal/ha treatment and significantly greater PP under the 10,000 Kg charcoal/ha treatment in the first measurement week than under the plots treated with 64,000 Kg charcoal/ha. There were no other statistically significant variations in PP between charcoal treatments in any other weeks. The variation in PP with charcoal treatment and measurement week can be seen in Figure 9.13.

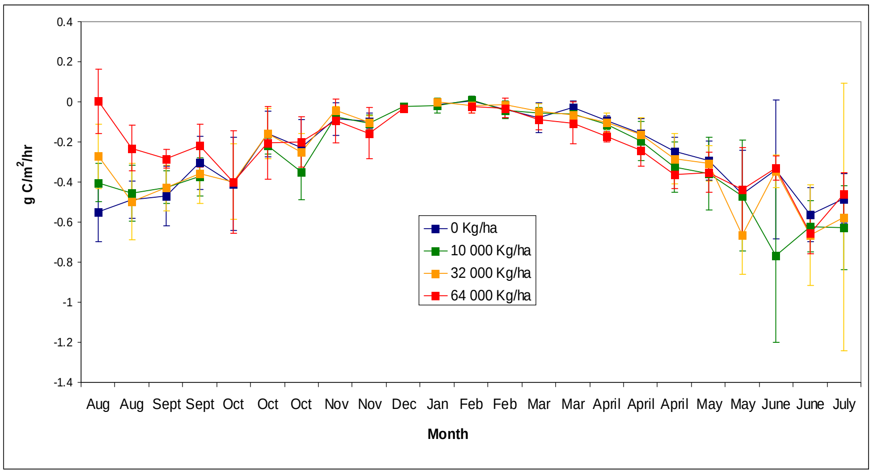

Figure 9.13 The variation in PP with charcoal treatment and measurement week, where error bars represent the standard deviation from the mean

9.3.3 Above-ground biomass stocks

A one-way ANOVA revealed no statistically significant difference in the total above-ground biomass produced under the plots treated with different amounts of charcoal.

Although not statistically significant, Figure 9.14 does reveal that the greatest total biomass was produced on the plots treated with the most charcoal (64,000 Kg/ha), and that all plots treated with charcoal produced greater total biomass over the trial period than the control plots.

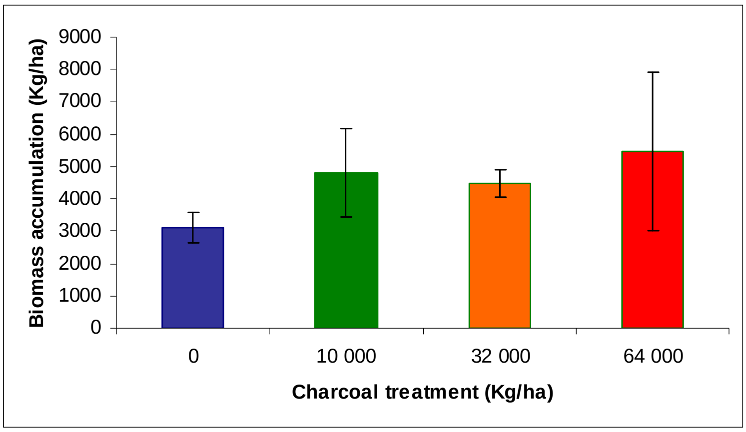

Figure 9.14 Variation in mean annual above-ground biomass accumulation with charcoal application. Error bars represent the standard deviation from the mean.

Further evidence to suggest greater above-ground biomass production under plots treated with charcoal comes from direct observation of the plots. Examination by eye in the months of October and November revealed grass of a healthier, greener appearance under the 64,000 Kg charcoal/ha plots than under any other treatments. The healthier appearance of these plots relative to the surrounding grassland can be observed in Figure 9.15 a. and b.

Figure 9.15 a. and b. Healthier grass growth under plots treated with 64 000 Kg charcoal/ha

9.3.4 Soil water chemistry

Soil water samples taken from the dipwells every fortnight over the one year trial resulted in the total collection of 141 samples for chemical analysis. Low water table levels in the drier summer months meant that water samples could not be collected on all visits to the site, and so not all months are represented in the analysis, with no samples collected in August 2009 or July 2010. As analysis of soil water DOC revealed that it was not normally distributed, it was log-transformed, and the log-transformed values used in all analysis. This was also true of soil water conductivity, soil water chloride concentration, and soil water nitrate concentration.

9.3.4.1 DOC

DOC concentration analysis was undertaken on 140 samples, with 35 from the control plots, 34 from the 10,000 Kg charcoal/ha treatment, 36 from the 32,000 Kg charcoal/ha treatment, and 35 from the 64,000 Kg charcoal/ha treatment. Statistical analysis via two-way ANOVA revealed that variation in date of measurement could explain 62.87% of the variation in DOC concentration, but that charcoal treatment was not significant. None of the covariates were able to explain any more of the variation in soil water DOC.

9.3.4.2 pH

Analysis of soil water pH was undertaken on a total of 141 samples. This consisted of 36 from the control plots, 34 from the 10,000 Kg charcoal/ha treatment, 35 from the 32,000 Kg charcoal/ha treatment and 36 from the 64,000 Kg charcoal/ha treatment. Statistical analysis via two-way ANOVA revealed that both charcoal treatment and date of sample collection had a significant effect on soil water pH. When soil water chloride concentration was included in the analysis, charcoal treatment continued to be a significant explanatory factor in soil water pH variation, with a combination of charcoal treatment, date of measurement and chloride concentration able to explain 80.07% of the variation. None of the other covariates included in the analysis could explain any of the further variation, and post-hoc analysis of the data revealed a significantly higher soil water pH below the 64,000 Kg charcoal/ha treatment than below the 32,000 and 10,000 Kg charcoal/ha treatments respectively. The main effects plot showing a greater soil water pH with charcoal application of 64,000 Kg/ha can be observed in Figure 9.16.

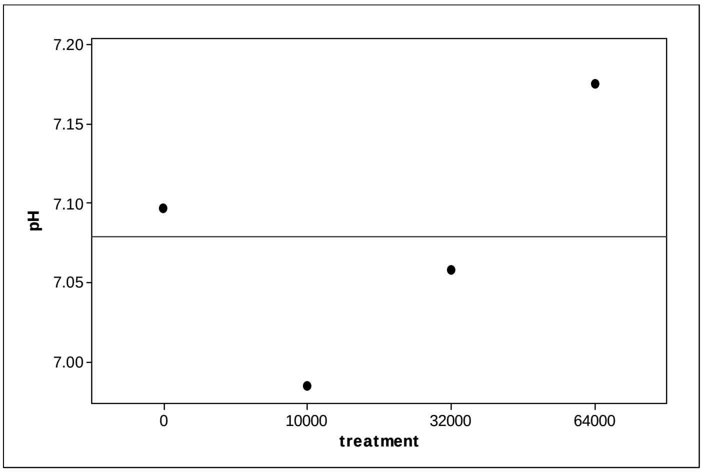

Figure 9.16 Main effects plot showing greater mean annual soil water pH under charcoal applications of 64,000 Kg/ha

9.3.4.3 Nitrate concentration

Soil water sample collection over the trial period resulted in the collection of 120 samples for the analysis of nitrate concentration. This consisted of 32 from the control plots, 28 from the 10,000 Kg charcoal/ha treatment, 32 from the 32,000 Kg charcoal/ha treatment, and 28 from the 64,000 Kg charcoal/ha plots. A two-way ANOVA undertaken on this dataset indicated that neither of these factors could explain any of the variation in nitrate concentration, and neither factor became significant with inclusion of any of the covariates.

9.3.4.4 Chloride concentration

A total of 131 soil water samples were analysed for chloride concentration, consisting of 34 samples from the control, 29 from the 10,000 Kg charcoal/ha treatment, 34 from the 32,000 Kg charcoal/ha, and 34 from the 64,000 Kg charcoal/ha treatment. Two-way ANOVA of the data revealed that both charcoal treatment and date of sample collection had a statistically significant effect on the concentration of chloride in soil water, together explaining 42.79% of the variation. Inclusion of the covariate DOC in the analysis revealed that its variation could explain some of the additional variation in chloride concentration; however with its inclusion the effect of charcoal treatment remained significant. Charcoal treatment, measurement week, and DOC concentration could together explain 46.91% of the variation in chloride concentration. Post-hoc analysis of the data revealed a significantly lower chloride concentration below the 10,000 Kg charcoal/ha treatment than the control.

9.3.5 Run-off water chemistry

Run-off water was collected on all fortnightly visits to the site if sufficient water had accumulated to enable analysis. This resulted in a total of 148 run-off water samples, however there was some inconsistency in the number analysed from each treatment due to inconsistent sample accumulation beneath plots. A normality test undertaken on run-off water electrical conductivity, DOC, chloride concentration, nitrate concentration and phosphate concentration revealed that these datasets were not normally distributed, and so they were log transformed, and the log transformed data used in all analysis of run-off water chemistry.

9.3.5.1 DOC

DOC analysis was undertaken on 136 run-off water samples, consisting of 30 from the control plots, 29 from the 10,000 Kg charcoal/ha treatment, 30 from the 32,000 Kg charcoal/ha treatment, and 47 from the 64,000 Kg charcoal/ha plots. Two-way ANOVA revealed that variation in date of sample collection could explain some of the variation in DOC, but that charcoal treatment was insignificant. Inclusion of the covariates chloride concentration and water table depth revealed that their variation could further explain some of the variation in DOC concentration, and in combination with date of sample collection 91.25% of the variation in DOC could be explained.

9.3.5.2 pH

pH analysis was undertaken on a total of 148 run-off water samples, consisting of 33 from the control plots, 33 from the 10,000 Kg charcoal/ha treatment, 30 from the 32,000 Kg charcoal/ha treatment, and 52 from the 64,000 Kg charcoal/ha treatment. Two-way ANOVA revealed that although variation in date of sample collection could explain some of the variation in run-off water pH, there was no significant variation explained by charcoal treatment. Inclusion of the covariate chloride concentration could explain some of the further variation in pH, and with its inclusion the variability in charcoal treatment became significant. This suggests that variability in chloride concentration between plots was masking the variation in run-off water pH explained by charcoal treatment. None of the other covariates could explain the additional variation in run-off water pH. Post-hoc analysis revealed a significantly greater pH for the 64,000 Kg charcoal/ha treatment than the 10,000 Kg charcoal/ha treatment. The main effects plot showing the adjusted mean run-off water pH over the trial period from each treatment is displayed in Figure 9.17, with a pH of 7.09 below the 10,000 Kg charcoal/ha treatment compared to a pH of 7.23 below the 64,000 Kg charcoal/ha treatment.

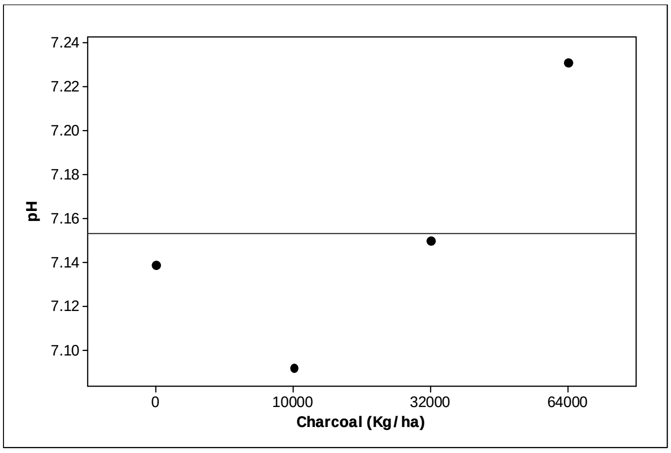

Figure 9.17 Main effects plot showing the variation in mean annual run-off water pH with increasing charcoal application

9.3.5.3 Nitrate concentration

Analysis of nitrate concentration was undertaken on a total of 125 run-off water samples. This consisted of 27 from the control plots, 30 from the 10,000 Kg charcoal/ha treatment, 25 from the 32,000 Kg charcoal/ha treatment, and 43 from the 64,000 Kg charcoal/ha plots. Two-way ANOVA revealed that variation in nitrate concentration could not be explained by charcoal treatment, but that variation in date of sample collection could explain 55.03% of the variation in nitrate concentration. Variation in none of the covariates was able to explain any of the additional variation in nitrate concentration.

9.3.5.4 Chloride concentration

Analysis of chloride concentration was undertaken on 138 run-off water samples, consisting of 30 from the control plots, 33 from the 10,000 Kg charcoal/ha treatment, 28 from the 32,000 Kg charcoal/ha treatment, and 47 from the 64,000 Kg charcoal/ha treatment. Two-way ANOVA revealed that variation in date of sample collection could explain some of the variation in chloride concentration, but that charcoal treatment was not significant. Although soil water conductivity was found to be significant it can not be said to explain the variation in chloride concentration due to their colinearity. Date of sample collection alone could explain 46.63% of the variation in run-off water chloride concentration.

9.3.6 Initial and final soil pH

Soil pH measured on samples taken from the plots before charcoal application ranged from a mean of 5.36 to a mean of 5.78, with the lowest value found below plots to be treated with 10,000 Kg charcoal/ha, and the highest below plots to be treated with 32,000 Kg charcoal/ha. One-way ANOVA revealed a significantly lower pH below the plots to be treated with 10,000 Kg charcoal/ha than below the un-treated plots, and the plots to be treated with 32,000 Kg charcoal/ha respectively. This variation in soil pH between plots prior to charcoal application is displayed in Figure 9.18.

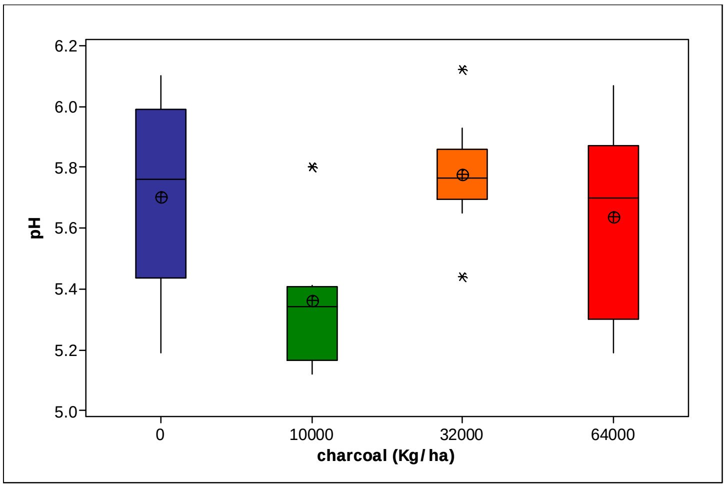

Figure 9.18 Pre-trial variation in soil pH between plots to be treated with charcoal

Soil pH measurements made at the end of the trial, one year after charcoal application, revealed a variation in pH from a mean of 5.15 below the 10,000 Kg charcoal/ha treatment, to a high of 5.48 below the 64,000 Kg charcoal/ha (Figure 9.19). One-way ANOVA undertaken on post-trial soil pH revealed a significant variation between charcoal treatments, with a lower pH below the 10,000 Kg charcoal/ha treatment than below the 64,000 Kg charcoal/ha treatment.

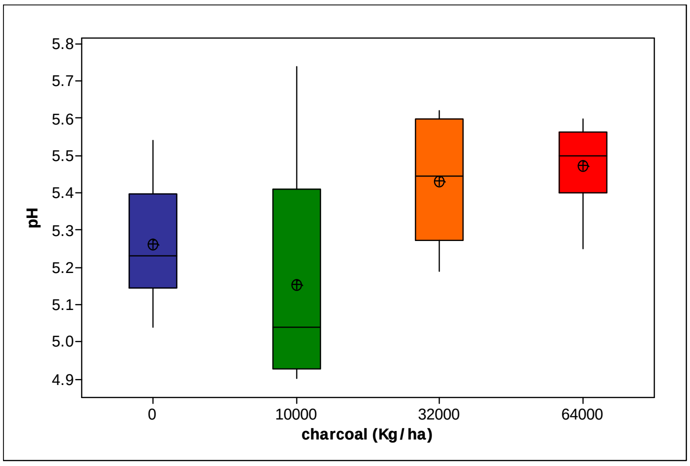

Figure 9.19 Post-trial variations in soil pH between different charcoal treatments

Comparison of Figure 9.18 and Figure 9.19 reveals that the significantly higher post-trial soil pH below the 64,000 Kg charcoal/ha treatment than the 10,000 Kg charcoal/ha treatment was not the result of charcoal application causing an increase in soil pH, but instead that soil pH has declined to a lesser extent under the maximum charcoal treatment.

9.4 Discussion

Greater PP under the 10,000 Kg charcoal/ha treatment than under the 32,000 and 64,000 Kg charcoal/ha treatments in the first measurement week is not unexpected due to the appearance of the pasture surface immediately after charcoal application (Figure 9.3 and 9.4). Although a significant difference with treatment was only measured in week 1, this was sufficiently large to cause the total mean measured PP beneath the 64,000 Kg charcoal/ha treatment to be significantly lower than that of the 10,000 Kg charcoal/ha treatment. The amount of charcoal applied caused a large percentage of the grass in these plots to be obscured from direct sunlight, and is therefore the likely cause of lower PP, due to limited photosynthesis. Despite variations in gaseous PP, measurements of total above-ground biomass growth showed no variation, indicating that treatment did not cause significantly less above-ground biomass production over the 1 year period. It is possible, although not known, that the higher PP flux under the control plots is the result of greater below-ground root production than that under the plots treated with large amounts of charcoal. Healthier grass production observed beneath the maximum charcoal treatment in the autumn months (Figure 9.15a and b) however suggests that charcoal application may have promoted PP, but that the delay in grass growth resulting from the inability of grass under this treatment to photosynthesise in the first month may be the reason for an insignificant difference in total above-ground biomass production over the one year period. Although not statistically significant, reference to Figure 9.12 suggests that PP under the 10,000 Kg charcoal/ha treatment was greater than that on un-treated plots. As such, the initial conclusion to draw from this trial would be that 10,000 Kg charcoal/ha should be applied directly to the surface of grasslands to increase PP, but that this will not result in observable increases in above-ground PP sufficient to encourage uptake by land-owners and farmers purely for crop productivity benefits. Application at this rate will make the Estate carbon neutral for approximately 32 years if applied as a one-off application. Although the visible appearance of grass under the maximum treatment implies that PP has the potential to improve with the highest application rates, this treatment level would not be recommended based on the results indicating reduced PP relative to the control over a one year period. If charcoal application was to be made as a one-off application, to make the Estate carbon neutral (as in the aim of this trial) for over 200 years, it is possible that increases in PP may continue, and the initial application effect would become progressively less important. To resolve these uncertainties longer trials are required where PP on land treated with 64,000 Kg charcoal is monitored for several more years, without the application of more charcoal, and compared to PP of un-treated land. The results relating to PP in this study do not support those of Spokas et al. (2009), Major et al. (2009) and Gaunt and Lehman (2008) in relation to increased PP under arable land-use, and until the necessary long term trials indicated here are undertaken, recommendation of charcoal applications as great as 32,000 and 64,000 Kg charcoal/ha can not be made. As indicated in Section 9.1 and 9.1.2 decisions to apply charcoal should not though be made solely on its impacts on PP, but instead on all aspects of the carbon balance, emphasising the need to assess the effects also on NER and NEE.

Although variation in PP was able to explain some of the variation in NER, the fact that charcoal treatment was a significant explanatory variable with its inclusion confirmed that NER was significantly greater under the 32,000 and 64,000 Kg charcoal/ha treatments than below the control plots, and that this was not the result of greater root and vegetation respiration. The higher flux thus suggests that these plots are experiencing either: greater microbial activity and a loss of native SOM in measurement weeks 6 and 23, or that the charcoal itself is decomposing and losing carbon to the atmosphere. The results imply that application rates of charcoal greater than 10,000 Kg/ha will cause a loss of carbon relative to if no charcoal had been applied, but that greater losses will not be experienced on a year round basis. Although the significant difference in flux is not a year-round phenomenon, the extent of the larger flux in the weeks when there was a significant difference means that mean NER over the entire measurement period is still significantly higher under the two greatest charcoal treatments. The results of MLR analysis (Table 9.1) suggest that the effect of temperature on NER is greatest under the plots treated with 32,000 Kg charcoal/ha. As microbial respiration is known to be affected by variations in temperature (Lloyd and Taylor, 1994), this suggests that microbial activity under the 32,000 Kg charcoal/ha treatment is more active than under all other treatments, and the control. Greater microbial activity under this treatment could help to explain the highest total NER from this treatment observed in Figure 9.9. Analysis of the MLR results does not though suggest that the same mechanisms are responsible for a high NER flux under the maximum charcoal treatment. Table 9.1 suggests that NER under the 64,000 Kg charcoal/ha plots is strongly affected by PP, suggesting that the measured NER is made up of root and vegetation respiration. With different results from the two highest charcoal treatments, and an assumption as to the source of NER, further research is needed, and no firm conclusions can be made. This analysis does not though help resolve whether any NER originating from greater microbial activity below the charcoal treated plots is the result of microbial respiration following decomposition of the native OM, decomposition of charcoal, or a combination of the two. According to the literature, decomposition of carbon within charcoal is very slow (Shindo, 1991), and with charcoal reportedly providing a suitable habitat for microbial biomass (Lehman and Rhondon, 2006), and a subsequent increase in native SOM breakdown, a source of NER from native SOM decomposition can be expected, although there is still a lot of debate relating to the stability of charcoal to warrant this also as a possible source of NER (Lehman et al., 2006; Nguyen et al., 2009; Knicker 2007). Despite uncertainties regarding the source of increased NER with charcoal application, an increased loss of carbon to the atmosphere has still been observed, suggesting that charcoal application may not result in the land to which it is applied being a net carbon sink, especially when the lower carbon flux from the atmosphere as PP is taken into consideration. This can not however be considered as an argument against charcoal application until the net affects on native SOM are known, and considered alongside the large amount of carbon contained within the charcoal applied to land.

Significantly greater NEE observed below the 10,000 Kg charcoal/ha treatment than the 32,000 and 64,000 Kg charcoal/ha treatments confirms that application of charcoal of amounts greater than 32,000 Kg/ha will result in a net release of carbon to the atmosphere. Until the source of increased carbon emissions can be identified, and the measured reductions in PP confirmed to be a long-term result of charcoal application, these factors should not be used as an argument against biochar application to land, however much further work is required before its application can be encouraged.

Analysis of the impacts of charcoal application on %SOC change following a year of treatment can help to resolve some of the uncertainties discussed, allowing firmer conclusions regarding its use and recommended application to be made. Significantly higher %SOC under the 32,000 Kg charcoal/ha treatment than the control plots one year after charcoal application initially implied that %SOC had increased with the spreading of charcoal. As the Walkley-Black method of %SOC analysis does not generally measure carbon contained in charcoal (Rowell, 1994) this implies that the increase was the result of SOC accumulation exceeding that of SOC mineralisation under the 32,000 Kg charcoal/ha treatment, and that the increase is in addition to the carbon sequestered directly in charcoal. A decrease in the %SOC of the control plots over the one year trial, and comparison of pre and post trial %SOC under each respective treatment, indicates that %SOC did not in-fact increase with charcoal application of 32,000 Kg/ha, but instead that %SOC under these plots and those treated with 64,000 Kg charcoal/ha did not decline, and was maintained. Comparison of the before and after trial %SOC reveals a decline in %SOC beneath the control plots and the 10,000 Kg charcoal/ha treatment. This analysis thus implies that SOC accumulation with charcoal application has not taken place, and that no additional carbon has been sequestered into soils as a result of application, but instead that the application of charcoal has maintained SOC levels, preventing their decline. This however was only true when greater than 32,000 Kg charcoal/ha was applied, with applications of 10,000 Kg charcoal/ha found to be insufficient to maintain native SOC stocks. As a result of this analysis it can be concluded that no additional carbon sequestration on top of that sequestered directly in charcoal can be claimed with any amount of charcoal spread on pasture. Nonetheless, soil carbon emissions to the atmosphere will be reduced with application of charcoal in the region of 32,000 Kg charcoal/ha, and the productivity and health of soil will be maintained (Lee et al., 2009; Lal, 2004), compared to un-treated plots or those treated with 10,000 Kg charcoal/ha, under which it will decline. Application of 64,000 Kg charcoal/ha, despite not reducing emissions, will not cause any increased losses of CO_2 to the atmosphere. Although a significant decline in the %SOC of plots treated with 10,000 Kg charcoal/ha relative to their %SOC one year previous, indicates that application at this small dosage is not sufficient to avoid carbon emissions to the atmosphere, it must be realised that there is not necessarily a net release of carbon to the atmosphere, as the carbon sequestered in the 10,000 Kg charcoal/ha must also be taken into account.

The results from %SOC analysis in combination with the carbon flux measurements can be used to help identify the source of higher NER below plots treated with the two greatest charcoal treatments. For %SOC to increase or be maintained under high charcoal applications, as has been witnessed in this trial, with a simultaneous reduction in total PP and no significant change in above-ground PP, it is not possible for the measured NER to have originated from native SOM. If PP under high charcoal treatments had been greater than under the control this could have been large enough to offset the losses as NER, with the products of PP being converted to SOC, as well as increases in root PP contributing to the SOC pool via root exudates (Rasse et al., 2005). As this was not observed some of the losses of carbon measured as NER from below the plots treated with charcoal must have resulted from decomposition of the carbon in charcoal, rather than decomposition of carbon in the native SOM pool.

In relation to the aims set out in Section 9.1.2, the results of this trial indicate that no immediate additional gains in carbon sequestration through changes in PP will occur within the first year of charcoal application to grazed pasture, but instead that PP will decline with applications in excess of 32,000 Kg charcoal/ha. Longer running trials are however required, as a potential for increased PP has been identified. With the source of higher NER below plots treated with greater than 32,000 Kg charcoal/ha thought to be charcoal decomposition rather than native SOM decomposition, the aim of ensuring that charcoal application does not enhance the flux of gaseous carbon from the soil to the atmosphere has also been met. Changes in %SOC indicate that charcoal application in excess of 32,000 kg/ha will maintain SOC levels compared to a deterioration and loss of SOC without application.

In addition to measuring the losses/gains of carbon via gaseous pathways it was also aimed to identify the impacts of charcoal application on the DOC concentration of soil and run-off water. The results of this analysis indicate that charcoal application will not be detrimental to the environment in terms of water quality deterioration, and will not promote SOC loss. This is in agreement with the conclusions regarding an insignificant change in native SOM decomposition with charcoal application discussed above. In relation to nutrient leaching no increase or decrease was found in nitrate and chloride loss from the surface of grasslands following charcoal treatment, again indicating that charcoal spreading on pasture will not be detrimental to water quality, nor will it result in losses of the applied fertiliser and economic costs which could accrue if this had occurred. The same results were true regarding losses of nitrate in soil water, however a lower chloride concentration was observed in soil water draining the plots treated with 10,000 Kg charcoal/ha. The reasons for a decrease in chloride concentration are currently unknown, however this decrease can not be considered as a negative impact, and suggests that water quality could be improved following charcoal application, although more research is required. The results showing an insignificant impact on nitrate loss following charcoal application to pasture are in disagreement with those from Chapter 7, relating to incorporation in arable soils, however the soils under study in Chapter 7 had been fertilised with a nitrate based fertiliser prior to the trial, compared to soils in this trial which were left unfertilised. Phosphate loss in both soil and run-off water was insufficient to enable comparisons with treatment. Greater soil and run-off water pH was observed below the 64,000 Kg charcoal/ha plots than below the control treatment, in agreement with the results from Chapter 7, indicating higher pH under higher charcoal applications. Although a greater soil pH was observed below the 64,000 Kg charcoal/ha plots at the end of the trial than beneath the control plots however, comparison of before and after soil pH reveals that this has not increased with application. This suggests that application of charcoal has maintained soil pH to the greatest extent, but that for soil pH to be increased, and for charcoal application to act as a liming agent, greater application rates will be required.

9.4 Conclusions

Lump-wood charcoal spread directly onto grazed pasture as a top-dressing in north east England has been shown in this trial to cause a reduction in annual above and below-ground PP at application rates >32,000 Kg/ha. Despite this decrease the appearance of grass in the autumn months is suggestive of potential increased PP, with the annual decrease the likely result of limited photosynthesis in the immediate weeks following application. If applied as a one-off process at such high levels, the increase in PP has the potential to continue and offset the short-term decrease, however more research is required.

In relation to carbon losses, significantly higher NER on plots treated with >32,000 Kg charcoal/ha indicates increased losses of carbon to the atmosphere with charcoal application, thought to be the result of charcoal carbon decomposition. These losses must be subtracted from the carbon sequestered in charcoal when calculating total carbon sequestration following charcoal application.

The application of 10,000 Kg charcoal/ha in this study did not have any detrimental effects (with no significant difference in PP, NEE or NER) relative to the control, suggesting that application of this material could advance, making the Wallington Estate carbon neutral for approximately 32 years.

No additional losses of DOC have occurred with any charcoal application rate in this trial, nor were losses of nitrate to soil and surface water enhanced or reduced. Issues regarding fertiliser loss from soils though require further research, as the field under study was un-fertilised. Although a higher pH was observed under soils treated with charcoal, this had not increased over the trial period, and greater application rates are required if charcoal is to be used as a liming agent.

All results reported here refer specifically to the consequences of lump-wood charcoal application to pasture, and can not be assumed to be representative of the effects of biochar produced from other sources and under different conditions, until more research specific to each is undertaken.

Chapter 10

Peatlands: The impact of afforestation on the total carbon and GHG budget

10.1 Peat at Wallington and its importance: an introduction

Peatlands, as carbon accumulating systems, are the most important of all terrestrial carbon stores, as although they cover only 3% of the global land area, they contain as much as 30% of total global terrestrial carbon (Gorham, 1991). This means that not only are the northern peatlands a very large carbon store, but that they also have the potential to be large carbon sinks/sources of carbon from/to the atmosphere depending on how they are managed or as climate changes. As it is difficult to envisage being able to control the effect of future climate change on the peatland environment and carbon flux, land-use change decisions are something over which we have greater control, and should therefore be used to reduce peatland carbon emissions, and increase terrestrial sequestration. Although peat covers only 5.89% of the Wallington Estate, it has the potential to be an extremely large sink or source of carbon depending on land-management. Research into peatland land-use at Wallington is essential, as although the peat is small in aerial extent it must be managed correctly to meet the aims outlined in Section 1.1.1, including that of sustaining current carbon stores in favourable condition, and enhancing the performance of all carbon sinks. The majority of this thesis has focused on the management of mineral soils, due to their major presence on the Wallington Estate (Figure 1.2), with some reference also to organo-mineral soils. The focus of this chapter is on the peat soils of the Estate.

A large soil sampling campaign undertaken at Wallington revealed the extent, distribution, and current condition of organic soils present on the Estate. In addition, the sampling campaign also revealed the land-uses under which this peat was located. These results, combined with reference to NSRI soil maps, indicated the presence of 3.24 Km^2 of peat, the location of which is displayed in Figure 1.5. The large majority of peat on the Wallington Estate is located under Harwood Forest Coniferous plantation (Figure 1.2), with small areas also in their pristine state or used for rough grazing. The focus of this chapter is on assessing the affects of afforestation on the carbon stocks within a peatland ecosystem.

10.1.1 Peat afforestation and the carbon cycle: a review of the literature

Pristine peatlands are carbon accumulating systems, with peat forming when water-logging suppresses the decomposition of OM (Montanarella et al, 2006). If this peat is left in an undisturbed state the water table remains permanently high, resulting in anaerobic conditions and slow peat decomposition (Minkkinen et al, 1999). The slow decomposition rate of pristine peatlands also results from peatland vegetation's high resistance to decay (Bubier et al, 1998). It is this slow decomposition, and the fact that it is exceeded by PP (Kivimaki et al, 2008) that results in peatlands in their pristine state being carbon accumulating ecosystems, with some studies suggesting net carbon sequestration rates as high as 70 $g/m^2/hr$ (Cannell et al., 1993), and others indicating rates of 10 to 40 $g/m^2/yr$ (McNeil and Waddinton, 2002; Cleary et al., 2005; Hendricks et al., 2007; Minkkinen et al., 2002).

One land-management change that has the potential to significantly alter the carbon balance of pristine peatlands is afforestation. Although forest plantation increases carbon sequestration in tree biomass and products (Kauppi et al, 2009), total terrestrial sequestration does not necessarily occur when trees are planted on peatland soils. During their growth phase trees accumulate carbon in their woody biomass; however factors associated with tree establishment can cause carbon to be lost from other parts of the ecosystem- e.g. peatland soils. A large amount of ground preparation is required if trees are to be planted on such soil types, with the major element being land drainage, a process which has greatly increased CO_2 losses from peat soils across the world (Holden, 2005; Funk et al., 1994; Bubier et al., 2003; Lloyd, 2006). Large areas of peatland have been drained for forestry, with 7% of Britain's peatlands having been afforested since 1945. Drainage for forestry is the most extensive of any peatland land-use management practice on a global scale (Minkkinen et al, 2002), and represents as much as 25% of peatland land-use change in Scotland (Ratcliffe and Oswald, 1988). The result of this drainage is an increase in the thickness of the aerobic surface layer- causing litter and peat decay rates to increase, due to aerobic peat decomposition occurring at a faster rate than anaerobic (Domisch et al., 1998). A 15 cm increase in the depth to the water table was observed by Alm et al. (1999) to cause an increase in soil respiration in the range 4 – 157 g C/m^2/yr. Lowering of the water table and extensive drainage is however vital, as it is necessary for tree establishment (Baker et al., 2007), and can significantly improve the growth of trees (Moore, 2002). Not only does drainage allow establishment, but it is also vital to prevent trees from suffering from windthrow as they age (Hargreaves et al., 2001). In addition to artificial drainage, water tables below afforested peatlands are lowered further as a result of water uptake by the trees themselves (Cannell et al., 1999).

Data suggesting increased carbon losses to the atmosphere with drainage are not however entirely conclusive. A change in the drainage of peat soils could result in changes in ground vegetation cover that could then increase carbon gains from the atmosphere (Dirks et al., 2000; Breeuwer et al., 2008). Other studies have also found either no change, or a decrease in carbon loss via respiration with lowering of the water table (e.g. Parmentier et al., 2008; Jauhiainen et al., 2005). This uncertainty exists when only drainage is considered, with the extent of uncertainty increasing further still when other effects associated with afforestation are considered in addition to drainage. Although there is a general consensus that peatland drainage increases decomposition, and converts the peatland in question from a sink to a source, the majority of research into drainage is in relation to drainage for agriculture (Domisch et al., 1998). The consequences of drainage for forestry are much less understood and uncertain (Worrall et al., 2010). There are some arguments that increased decomposition will be offset by a decrease in peat temperature (Hytonen and Silfverberg, 1991; Minkkinen et al., 1999), litter quality and peat pH (Laine et al., 1995), implying therefore either no change in decomposition as a result of drainage, or only a small increase.

With soil respiration being only one aspect of the peatland carbon balance, other carbon loss/gain pathways which may be affected by afforestation must also be assessed. The impact of peatland afforestation on DOC loss was measured by Sallantaus (1994, Cited in: Minkkinen et al., 1999), who suggested slight increased carbon leaching following drainage. Equally, Grieve and Marsden (2001) reported an increase in DOC concentration in soil solutions from a forest compared to a moorland. Contradictory evidence though was provided by Hope et al. (1994), who found lower losses of DOC from forests. Clearly there is a large amount of uncertainty within the literature relating to carbon loss/gain following the afforestation of pristine peatlands, indicating the need for much further in-depth research.

10.1.2 Chapter aims

This chapter attempts to identify peatland land-use which should be undertaken on the Wallington Estate, in order to preserve the current high carbon stock, and prevent its release to the atmosphere. The impact of land-management on the peatland carbon cycle has been widely studied, and so the aim of this chapter was to combine the results of literature with direct experimental evidence, to investigate whether organic soils at Wallington are being managed in a way which will preserve their natural high carbon stocks, or if they are being mismanaged, resulting in carbon emissions to the atmosphere. As the majority of peat on the Wallington Estate is located under forestry plantation, the objective was to compare the carbon flux from afforested peatland with that from peat in its pristine state, to identify whether trees should be removed, and the peatlands be allowed to return to their pristine state, or whether peatland afforestation should continue.

In line with the aim of this thesis to assess the impacts of land-use on the total land carbon stock (i.e. the above-ground and below-ground carbon stocks) it was necessary to consider the carbon gains that would be achieved in the biomass of coniferous forest trees located on any afforested peat, in addition to studying the impacts on SOC.

The initial aim of the study was to assess the impact of peatland afforestation on NER and DOC loss from the peat and in soil water. Measurement of DOC loss was deemed important not only for a measure of total carbon loss from the land, but also to meet the aims outlined in Section 1.1.2 regarding the impacts of land-use change on water quality. Peat NER fluxes and soil water DOC concentrations would be compared between land uses, one under afforestation, in the context of changes in water table, soil solution chemistry and surface-air temperature. It was then hoped to establish an annual flux estimate for both of these parameters for both afforested and pristine peat.

With a measurement of NER and DOC flux from each land-use, the objective was to then compare the extent to which these carbon losses would contribute and affect the total carbon and total GHG budgets of peatland systems. Whilst not measured in this study it was realised that peatland land-use change could impact on other forms of carbon loss/ gain and GHG emissions/sequestration. It was believed that any assessment of land-use change should consider the effect on PP, POC, Dissolved CO_2, and $CH_{4,}$ in addition to NER and DOC. If increased NER and DOC were found from the afforested peatland relative to the pristine peatland, the objective then was to identify if the gains in carbon resulting from afforestation, and gains in tree biomass could be large enough to offset the losses from peat.

10.2 Approach & Methodology

10.2.1 Study site

The study took place on the NT's Wallington Estate in Northumberland, north east England (Figure 1.1), on an area of peat known locally as Greenleighton Mire. The afforested peat is on UK Government Forestry Commission land located within Harwood Forest (55°10' N, 2°3'W). The site experiences mean annual temperatures of 7.6°C and annual precipitation of 950mm (Mojeremane et al, 2009). Establishment of the forest involved peatland drainage with open ditches spaced at approximately 20-30m. Trees were established on mounds placed at approximately 2m intervals. The trees occupying the area of peat are Sitka Spruce planted in 1981 (Figure 10.1), and have a yield class of 10 (Forestry Commission data). Prior to forest establishment the land was used for grazing domestic animals.

Figure 10.1 Sitka Spruce trees planted on peatland at Wallington

The other part of the study site is immediately adjacent to the forested mire, but located outside the boundary of Harwood forest. This land is fenced off from grazing animals and hence treated as a "pristine" peatland in this study. Grazing and a small amount of agricultural improvement are however known to have taken place in the past, with the water table having been lowered by drainage, although these drains are now blocked.

Trampling cattle are also reported to have caused damage to mosses prior to 1995. The vegetation occupying the pristine peatland is typical of that of pristine peatlands, with a variety of mosses and sedges, including abundant sphagnum moss (eg. Sphagnum papillosum) and Cranberry (Vaccinium oxycoccos), with Hare's tail cotton grass (Eriophorum vaginatum) and bog rosemary (Andromeda polifolia) also present, (Hewins et al., 2001) (Figure 10.2).

Figure 10.2 Pristine peat on the Wallington Estate

Although vegetation indicative of healthy pristine peatland such as bog rosemary (Andromeda polifolia) is present, the occurrence also of Hare's tail cotton grass (Eriophorum vaginatum) and purple moor grass (Molinia caerulea) indicate that the peatland is not in a perfect pristine state, and may have suffered from burning in the past (Wallington Biological survey). Soil type beneath both land-uses is classified as Winter Hill- a deep peat, greater than 40cm in depth, therefore fitting the classification of "deep peat" as ascribed by the Soil Survey of England and Wales (Avery, 1980). Both land-uses are located at an altitude of 265m. The fact that altitude was controlled in the experiment meant that although altitude may be responsible for variations in peatland NER, it can not be used to explain any variability in this study.

10.2.2 Peat and surface vegetation NER

To enable measurement of NER from the peat surface six PVC collars were inserted into the ground under both the forested and pristine sections of Greenleighton Mire (Figure 10.3a and 10.3b). The collars were inserted to a depth of approximately 5cm, leaving 5cm protruding from the peat surface. These were left in place for 2 weeks before any measurements were made, to ensure that measuring CO_2 flux from root death caused by insertion of the collar would be minimised. The collars remained permanently located in the ground for the duration of the trial.

Figure 10.3a

Figure 10.3b

Soil respiration collars installed under a. afforested peat; b. pristine peat

Under the forest the collars were placed on the mounds rather than the ditches, as the ditches were often waterlogged and would have made measurement difficult. Under both land-uses the collars were randomly located so as to not introduce any bias into the experiment. NER was monitored on a fortnightly basis from the soil surface using a dynamic dark closed chamber and infra-red gas analyser as described in Section 4.2.4.1. This allowed the change in CO_2 concentration within the chamber to be measured, and hence the CO_2 efflux from the soil surface. Analysis took place between the hours of 13.30 and 15.30 on the day of initial measurement, and subsequently every measurement period following. This was in an attempt to minimise any effects associated with daily temperature, moisture and light variation which could obscure the results due to 24 hour variability in soil carbon flux (Mielnick and Dugas, 2000). Measurements were taken on a fortnightly basis for the entire trial period of 52 Weeks.

10.2.3 PP

No measurements of PP were made in this study. Although PP of the peat and surface vegetation could have been measured using the methodology described in Section 4.2.4.3, it was decided that this would not be an accurate reflection of the impact of peatland afforestation on PP, as this method could not measure the PP of the major aboveground vegetation on one of the sites, i.e. the trees. Estimates of gains in PP under afforested and pristine peat in both tree and peatland ground vegetation were achieved instead by reference to literature, as will be described in Section 10.2.8 and Section 10.2.9.

10.2.4 Water chemistry analysis and water table depth

Concentrations of DOC were measured on soil water collected from dipwells located alongside each respiration collar, under both types of peatland land-use. Measurements of soil water DOC concentration were made from a total of six dipwells below the forest, and six dipwells from the pristine peatland system. Dipwells were constructed as in Section 4.2.6 and were inspected on a fortnightly basis following NER measurement, with water samples withdrawn for DOC analysis. Water samples collected were filtered to 0.45 µm, and DOC concentration analysed using the method of Bartlett and Ross (1988), described in Section 4.2.6. Electrical conductivity and pH were analysed immediately in the lab on the filtered samples using electrode methods. On all visits to the sites depth to the water table was measured as described in Section 4.2.6.

10.2.5 Surface-air temperature

Measurements of surface-air temperature at the time of each IRGA reading were required for calculation of NER, as indicated in Equation 4.1. A temperature sensor was therefore installed within the chamber, allowing surface-air temperature to be recorded along with every reading of CO_2 taken over each 124 second IRGA reading. In addition to allowing calculation of NER, surface-air temperature was also measured as it was a variable over which there was no control, and was hence required to be included as a covariate in all statistical analysis.

10.2.6 Statistical analysis to identify variations with land-use

The experimental design in this chapter represents a two-factor design with the factors being land use (afforested vs. pristine) and time (week/month of measurement). As an initial approach ANOVA was used to compare the impacts of afforestation on NER and DOC, with respect to natural and seasonal variations. Subsequently ANCOVA was used to establish if these land-use controls remained significant even allowing for uncontrolled experimental variables. Covariates measured and used in the analysis were soil water pH, surface-air temperature, and water table depth. Before any analysis was undertaken it was ensured that all data was normally distributed. Any data that did not meet this requirement was log-transformed and re-tested for normality. If the response variable in the analysis was log-transformed then 1/temperature was used as covariate, compared to temperature when assessing the variation in an un-logged response. Any significant differences identified by ANOVA (or ANCOVA) were then post-hoc tested using the Tukey test, to identify where significant differences occurred between land-use or levels of the factor time.

10.2.7 Annual ground surface NER calculations

With the impact of peatland afforestation on NER established, the question then was how best to predict the annual NER flux from each respective land-use. The results from ANCOVA will indicate which factors and covariates can best explain the variation in ground surface NER from peatlands, however these results will have been achieved from analysis of the combined afforested and pristine peatland datasets. The annual flux from each land-use type, if this is significantly different with land-use, will be best achieved knowing the respective controls upon NER under each land-use. The combined afforested/pristine peatland datasets were therefore split into datasets from afforested and pristine peatlands respectively, and MLR analysis undertaken to identify the variables responsible for explaining the greatest amount of variation in NER, under each respective land-use.

With the controls on NER established for each land-use, an annual estimate can be made for both the afforested and pristine peat as long as these covariates are known across the annual cycle. As the temperatures recorded alongside NER measurements in this chapter were taken between the hours of 13.30 and 15.30 they are unlikely to be representative of mean daily temperature for the respective month, due to diurnal variation. During investigation into the impact of land-use on ground surface NER, the calculated hourly fluxes can be compared, as all land-use ground surface NER fluxes were measured over the same time period, however when calculating annual fluxes, temperatures representative of mean monthly values are required. A lack of monthly average temperature data specific to the Wallington Estate meant that data from the nearest accessible weather station was required. Temperature data was thus obtained from the Durham University weather station and is displayed in Table 8.1 (www.geography.dur.ac.uk/projects/weather/Home/tabid/666/Default.aspx)

In relation to other controls on NER and DOC, the measured variables were not considered to undergo diurnal variation, therefore the mean values calculated on each visit to the site were considered to be representative of mean monthly values, and these values were used in calculations of annual ground surface NER flux.

10.2.8 Estimating the impact on the total carbon and GHG budgets: 1

As explained in the introduction, it is important to not only consider NER and DOC loss/gains resulting from land-management interventions, but also the impacts on all other uptake and release pathways of the carbon and GHG budgets. As with many previous studies however, (see: Worrall et al., 2010), this study was unable to assess the impact on all of these factors due to time limitations and equipment shortages, leaving a need to estimate the impact on these fluxes by other methods. The impact on the total carbon and GHG budget was therefore assessed by combining the results here with those results from other studies. A review of the literature revealed that no full carbon budgets have been established for any afforested peatland sites in the UK, or in any boreal or sub-boreal region (Worrall et al., 2010), therefore the results for individual components of the budget from individual studies must be combined. This chapter uses the Bayesian meta-analysis approach of Worrall et al. (2010) whereby individual results for individual components of the carbon or GHG budget of a peat soil can be compared with results from other studies of the same or other components of the carbon or GHG budget. In this way, studies that are, of necessity, incomplete, can be used in a wider context. Results from separate components of the carbon or GHG budget can be compared by reference to the stoichiometry of the carbon or GHG budget. Using the stoichiometry of the carbon budget proposed by Worrall et al. (2009) the components of the carbon budget investigated were combined with those from other studies to reveal the affect of afforestation on the total carbon budget. Worrall et al. (2009) state that

$$100C_{pp} \Rightarrow 35C_{R} + 26C_{DOC} + 4C_{DisCO2} + 9C_{POC} + 22C_{RES}$$ **Equation 10.1**

Where: pp = primary productivity, R = net ecosystem respiration, DOC=dissolved organic carbon; CH_4=methane; disco$_2$=dissolved CO_2; POC = particulate organic carbon; and RES = residual carbon stored in the soil.

Equation 10.1 indicates that the two aspects of the carbon budget investigated in this study are important in terms of their overall weighting in the carbon budget. In terms of the effect of afforestation on the total GHG budget the following stoichometric equation (Equation 10.2) can be used.

$100C_{pp} \Rightarrow 35C_{R} + 10C_{DOC} + 96C_{CH4} + 4C_{dissCO2} + 4C_{POC} + 22C_{Res}$ **Equation 10.2**

Where : pp = primary productivity, R = net ecosystem respiration, DOC=dissolved organic carbon; CH_4=methane; $dissco_2$=dissolved CO_2; POC = particulate organic carbon; and RES = residual carbon stored in the soil.

Using these equations and the literature reviewed in Worrall et al. (2010) the impact of afforestation on the total carbon budget and GHG budgets were estimated.

10.2.9 Estimating the impact on the total carbon and GHG budget: 2

Although the results produced by the method described in Section 10.2.8 will provide both a probability of improvement in the peatland carbon budget following afforestation, plus a variance estimate on that probability, the method of combining information will not reveal the magnitude of improvement/worsening of the aspect of the carbon budget in question. In order for the NT to gain a more accurate estimate of the impacts of peatland afforestation/deforestation in relation to their current carbon and GHG emissions, a method to acquire a quantitative value of the carbon sequestration/emission possible following such land-use change is required. To attempt to gain an improved estimate of the relative impacts on various aspects of the carbon and GHG budget the annual carbon losses via NER estimated in this study can be compared with the annual carbon gains estimated in other studies as PP. Although quantitative values for losses/gains in carbon from other studies regarding dissolved CO_2, POC and CH_4 should also be included in any estimates of the annual carbon/GHG balance in line with Equations 10.1 and 10.2, a lack of data meant that such information could not be included in this methodology. As this study measured the NER of peatland soils, ground surface vegetation, roots, and leaf litter, a value for leaf litter addition to the soil was also required. Litter trays were therefore installed alongside soil respiration collars under the afforested peat as described in Section 8.2.4, and leaf litter collected and weighed following sufficient accumulation as in Section 8.2.4.

10.3 Results and discussion

10.3.1 The impact of afforestation on peat ground surface NER

Measurement and observation over a 12 month period resulted in a total of 25 x 6 NER measurements for the pristine peatland, and 23 x 6 measurements for the afforested peatland, the smaller sample size from the afforested peatland being the result of an equipment failure at a time of extreme temperature. These measurements yielded a mean ground surface NER flux from the pristine peatland of 0.042 g $C/m^2/hr$ (median 0.029 g $C/m^2/hr$), and a mean NER flux from the afforested peatland of 0.079 g $C/m^2/hr$ (median 0.065 g $C/m^2/hr$). One-way ANOVA revealed a statistically significant difference in ground surface NER between land-uses; however land-use alone could only explain 8.69% of the variation in NER. Comparison of the mean and median flux values between land-uses suggests that afforestation results in a greater NER flux from the ground surface of peatlands. One-way ANOVA does not though indicate whether the greater flux from afforested peat is a year round occurrence, or if there are some months in which the difference is not significant, or in-fact reversed. A two-way ANOVA was thus undertaken, revealing that variation in both week of measurement and land-use could explain some of the variation in ground surface NER. To enable this comparison however, week 8 and 12 had to be removed from the dataset, leaving a total of 23 x 6 measurements for both land-uses. Together the variation in these factors could explain 72.47% of the variation in ground surface NER. When the interaction between week and land-use was also included in the GLM the predictive power of the model increased to 78.07%, with a significant interaction evident between land-use and week. Only in week 44 and 46 (October) was there found to be a significant difference in ground surface NER between land-uses, with a significantly higher flux from the afforested peat in these weeks. This variation in NER with land-use and season is displayed in Figure 10.4.

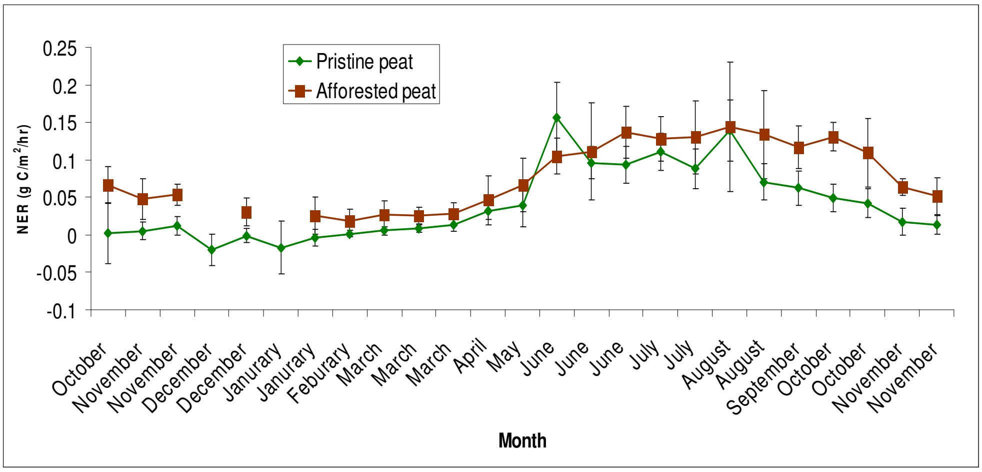

Figure 10.4 The variation in NER with land-use and season, where coloured symbols represent the mean NER and error bars represent the standard deviation

Further analysis with the inclusion of covariates was then undertaken, allowing establishment of the impact of land-use on ground surface NER when uncontrolled variables in the experiment were held constant. Log transformation did not improve the distribution of any of the variables, therefore the un-transformed variables were used in all analysis discussed here. With surface-air temperature included in the analysis, land-use, measurement week, and the interaction between measurement week and land-use remained significant, with surface-air temperature unable to explain any of the further variation in NER. To enable the controls on ground surface NER to be better established, and make the results of the analysis more applicable to other studies, it was decided that month rather than measurement week should be used as a factor in the analysis. This confirmed a significant interaction between time of year and the impact of land-use on NER. Post-hoc comparison revealed a significantly higher ground surface NER flux from the afforested peat in only October and November. When month and land-use were included in the model as opposed to week and land-use, the predictive power of the model decreased slightly, with an r^2 value now of 70.88%. Inclusion of the covariate surface-air temperature in the analysis revealed that variation in surface-air temperature was able to explain some of the further variation in NER, with land-use and month both accounted for, with variation in the combination of factors and covariates now able to explain 73.03% of the variation in ground surface NER. Variation in land-use and month continued to be able to explain some of the variation in NER, confirming that peatland land-use in October and November does have a significant affect on ground surface NER. Inclusion of depth to water table as a covariate in the analysis could not explain any more of the variation in NER; however the affect of land-use and month remained significant with its inclusion, revealing that differences in water table with land-use and month were not causing the significant observed differences in ground surface NER.

The inclusion of further covariates in the analysis required investigation on a smaller dataset, as soil water electrical conductivity, soil water pH and soil water DOC were only measured for the months June to November due to equipment availability. On this smaller dataset variation in month, land-use, and the interaction between month and land-use could all explain some of the variation in ground surface NER, however this amounted to only 58.10% of the variation. Including depth to water table increased the predictive capacity of the model to 59.74%, and water table was able to explain some of the further variation in NER, with land-use, month and the interaction between land-use and month remaining significant with its inclusion. Water table depth could therefore account for more of the variation in ground surface NER for a particular month and land-use, but was not responsible for the differences in ground surface NER between land-use. When soil water pH and soil water electrical conductivity were included in the analysis they were found to have an insignificant affect on NER, and land-use and month remained significant with their inclusion, suggesting that the evident differences in ground surface NER with land-use are the result of land-use, and not the impact that land-use is having on soil water chemistry. The inclusion of surface-air temperature was unable to explain any of the further variation in ground surface NER, which does not agree with the results from the larger dataset, however the affect of land-use, month and the land-use/month interaction again remained significant. This indicates that surface-air temperature can not explain the variation in ground surface NER beneath a constant land-use and on a particular date, but that when temperature was held constant there were still significant differences between land-use, implying that the differences in ground surface NER caused by land-use were not in-fact the result of temperature differences caused by the land-use itself.

10.3.2 The impact of afforestation on DOC

Sample collection, and hence analysis of results on the effect of afforestation on soil water DOC concentration was undertaken for the months June to November. This resulted in a total of 10 x 6 samples from the pristine peatland, and 10 x 6 samples from the afforested peatland. A mean DOC concentration of 198.0 mg/l was measured from the afforested peatland, compared to a mean of 126.2 mg/l from the pristine peatland. To enable the effect of afforestation on DOC to be established, one-way ANOVA was undertaken on these DOC concentration values. A significant difference with land-use was observed, in agreement with a study by Wallage et al. (2006), which found greater losses of DOC from drained compared to un-drained peat. Land-use however, could only explain 20.05% of the variation in DOC concentration, and one-way ANOVA did not allow time of year and the interaction between land-use and time of year to be assessed. Nor did the analysis include any of the uncontrolled covariates in the experiment which may be masked or be masking the affect of land-use. A two-way ANOVA was thus undertaken; with variation in land-use and week both able to explain some of the variation in DOC concentration, although there was no significant interaction between land-use and week. Inclusion of land-use and measurement week in the analysis could explain 52.04% of the variation in DOC concentration, but the significant difference between land-use was not evident for any individual measurement week, and was only significant when comparing mean DOC concentrations from the entire measurement period. The difference in soil water DOC concentration between afforested and pristine peat can be observed in Figure 10.5.

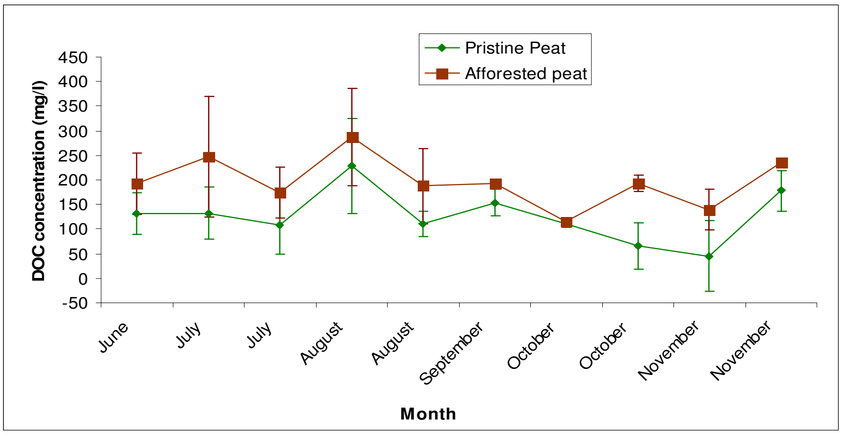

Figure 10.5 The variation in DOC concentration with peat land-use, where coloured symbols: mean NER; error bars: standard deviation from the mean

ANCOVA undertaken on the dataset indicated that variation in surface-air temperature could not explain any of the further variation in DOC concentration under a constant land-use or week, and with its inclusion land-use and week remained significant controls. As in the analysis of NER, log-transformation did not improve the distribution of any variables, and so the un-transformed values were used in all analysis of DOC. This indicates that there are not significant differences in temperature caused by land-use, supporting the results from the ground surface NER analysis. The same results were observed with the inclusion of all other covariates; depth to water table, soil water pH and soil water electrical conductivity.

As with the ground surface NER data, it was decided that month rather than measurement week should be used as a factor in the analysis, if the results of this study are to be applied to other land-use change scenarios. Variation in both month and land-use could explain some of the variation in DOC concentration; however there was no significant interaction between month and land-use. Analysis by month, rather than week of measurement, indicated that only 30.20% of the variation in DOC concentration could be explained, compared to 52.04% when analysed by measurement week. ANCOVA with depth to water table included as a covariate revealed that water table depth could explain some of the further variation in DOC concentration; however with its inclusion the effect of land-use became insignificant. This indicates a difference in water table depth between land-uses that is responsible for the difference in DOC concentration. None of the other covariates surface-air temperature, soil water pH, or soil water conductivity could explain any further variation in DOC concentration with land-use and month accounted for.

10.3.3 Predicting the total annual peat NER budget of afforested and pristine peatlands

A lack of soil water chemistry measurements for the months December to May meant that these variables could not be included in any regression analysis and annual ground surface NER flux estimate calculations. Although month of measurement was able to explain the greatest amount of variation in ground surface NER under afforested peat, the regression equation produced in this analysis could not be used to accurately estimate annual ground surface NER, as this would assume that the measured NER values recorded in this study were representative of mean monthly ground surface NER flux (see Section 10.2.7). Regression analysis revealed that 35.30% of the variation in ground surface NER could be explained by surface-air temperature, with water table depth unable to explain any of the variation. No improvement in the distribution of data, or the r^2 value of regression analysis following log-transformation meant that all non-transformed variables were used. Equation 10.3 produced by the regression analysis was then used to calculate an annual NER flux estimate from afforested peatland at Wallington, using the mean monthly temperatures displayed in Table 10.1.

$NER = 0.020 + 0.0052\,t$ **Equation 10.3** $r^2 = 35.3\%$; n = 128

Where NER: g $C/m^2/hr$; t: surface-air temperature (°C)

As with ground surface NER flux estimates for afforested peat, soil water pH, electrical conductivity and DOC concentration could not be used in estimates of annual ground surface NER flux from pristine peat in this study. The same issues regarding month of measurement also meant that this variable could not be used to predict annual ground surface NER from pristine peat. Regression analysis revealed that 70% of the variation in pristine peat NER could be explained by variation in a combination of water table depth and surface-air temperature. Equation 10.4 produced by this regression analysis was then used to calculate an estimate of annual NER from pristine peat at Wallington, using the mean monthly temperatures and mean water table depths displayed in Table 10.1.

$NER = -0.038 + (0.0015wd) + (0.0058t)$ **Equation 10.4** $r^2 = 70\%$; n = 107

Where: NER: $gC/m^2/hr$; wd: water table depth (cm); t: surface-air temperature (°C)

An estimate of the annual NER flux from the afforested and pristine peat is presented in Table 10.1.

Month	Mean temperature (°C)	Afforested NER (g C/m^2/month)	Mean WT depth(cm) (pristine peat)	Pristine NER (g C/m^2/month)
January	3.4	27.96	0.5	0.00
February	4.7	29.77	1.75	0.00
March	6.8	41.04	3.76	4.90
April	8.9	47.53	5.67	15.45
May	11.3	58.35	12.2	33.32
June	13.5	64.65	10.39	39.45
July	15.4	74.12	5.33	43.40
August	15.8	75.65	6.08	45.92
September	13.6	65.02	11.17	40.69
October	9	49.50	8.92	19.92
November	6.5	38.60	1.8	1.44
December	3.6	28.73	-0.75	-13.69
Annual NER (g C/m^2)		**600.91**		**230.81**

Table 10.1 Annual NER estimates from afforested and pristine peat at Wallington, and the data and calculations used in the production of these estimates, where WT: water table

As measurement of DOC was only undertaken during the months June to October in this study, and as measurements relate to DOC concentration and not DOC flux, an annual estimate of DOC flux has not been calculated for either land-use.

10.3.4 The impact of afforestation on the total carbon and GHG budgets

As explained in Section 10.1.2, a further aim of this study was to combine the results presented here with the impacts on other carbon flux pathways presented in previous studies. The reasoning behind this was to achieve the objective of establishing the consequence of peatland afforestation on the total annual peatland carbon and GHG balance.

The results of this study show an increase in ground surface NER with afforestation relative to pristine peat, therefore a worsening of the NER aspect of the carbon budget. The study also reveals an increase in DOC concentration with afforestation relative to pristine peat, a worsening also of that aspect of the carbon budget. These results were then added into the methodology used in Worrall et al. (2010), to estimate the impact of peat afforestation on the total carbon and GHG budgets. Although the results of peatland afforestation in this current study suggest that a land-use change from pristine peat to afforested peat will be detrimental to the environment in terms of increased carbon emissions from land to atmosphere, this does not take any account of the other forms of carbon and GHG benefit that may simultaneously accrue as a result of the land-use change.

Of 13 studies, in peer-reviewed journals, into the impact of afforestation on the NER flux from sub-boreal peatlands, identified in Worrall et al. (2010), none were found to cause an improvement (where improvement is considered relative to the atmosphere and would mean a decreased flux of GHG to the atmosphere) in that particular aspect of the carbon budget. This study can be grouped with the 13 previous studies, to reveal zero studies out of a now total of 14 showing an improvement in the NER component of the total carbon budget. With respect to DOC flux, the results of this study can be added to those considered in Worrall et al. (2010) to reveal that out of 3 studies assessing the impact on DOC, none were found to show an improvement in the DOC aspect of the total carbon budget. Combining the results of this study with the literature, and weighting the components of each budget according to Equations 10.1 and 10.2, reveals a 77% probability that peatland afforestation will result in an improvement of the carbon budget, and a 91% probability that peatland afforestation will result in an improvement of the GHG budget. Despite the increased losses of carbon via ground surface NER and DOC due to peatland afforestation, this study, combined with the results from previous studies in the literature, suggests that peatland afforestation is beneficial to the environment, and will increase the sequestration of atmospheric carbon. This high probability of sequestration following peatland afforestation, despite the detrimental effects upon ground surface NER and DOC, is the result of an increase in PP and a decreased methane flux, achieved by the planting of trees. The weighted equivalent number of complete carbon budgets with this new data included now increases to 8.99, and the equivalent number of complete GHG budgets to 8.85. The variance on the probability estimate of carbon and GHG benefit was calculated as in Worrall et al. (2010) using Equation 10.5, revealing a likelihood of an improved carbon budget following afforestation of 77%, with a variance of 0.02%, and a likelihood of an improved GHG budget of 91%, with a variance of 0.02%.

$$\text{Var} = \frac{\alpha\beta}{(\alpha + \beta)^2 (\alpha + \beta + 1)}$$ **Equation 10.5**

Where α = no. of studies with an improvement, β = total number of studies with no improvement

10.3.4.1 A quantitative value for the impacts of afforestation on the carbon and GHG balance

In this study the ground vegetation was not removed from the collars, thus the measured values for ground surface NER include losses of carbon to the atmosphere from root respiration and ground vegetation respiration in addition to peat decomposition. With these accounted for, the forested peat is annually losing 370.1 g C/m^2/yr more than the peat in its pristine state, implying that for the forested peat to be a carbon sink a flux from the atmosphere of at least 370.1 g C/m^2/yr is needed. As no measure of this flux was made in this study, a literature review was undertaken to identify if this could be true of forested peatlands typical of the UK.

In relation to the pristine peatland, the ground surface NER value of 230.31 g C/m^2/yr refers to all losses to the atmosphere from the ground vegetation, peat decomposition and root respiration. To calculate the NEE of the pristine peatland a value for PP of ground vegetation and roots is also needed. Average PP values of peatland ground vegetation and roots from the literature are presented in Table 10.2, column 2.

	PP (g $C/m^2/yr$)	NER (g $C/m^2/yr$) (From current study)	NEE (g $C/m^2/yr$)
Clay (2009)	-181.65	230.31	48.66
Rowson (2007)	-127	230.31	103.31
Worrall et al (2009)	-246	230.31	-15.69
Clymo et al (1998)	-189.4	230.31	40.91
Garnett (1998)	-229	230.31	1.31
Mean carbon flux (g $C/m^2/yr$)	**-194.61**	**230.31**	**35.7**

Table 10.2 Pristine peat carbon flux values and calculations using results from this current study and previous studies in the literature

Assuming the lowest of these figures is true for PP at Wallington, the pristine peat would have a NEE of 103.31 g $C/m^2/yr$. Assuming the highest of these figures is true for PP at Wallington, the pristine peat would have a NEE of -15.69 g $C/m^2/yr$. These figures suggest that at the maximum rate of PP quoted in the literature, the pristine peat at Wallington will be a small sink of carbon, but at the lowest rate a source of carbon to the atmosphere, with a mean emission of 35.7 g $C/m^2/yr$. In relation to the forested peatland the ground surface NER value of 600.91 g $C/m^2/yr$ refers to all losses from the ground to the atmosphere from ground vegetation beneath the forest, peat decomposition, litter decomposition and the roots of ground vegetation and trees. To enable calculation of the NEE of the forested peat a value for PP of the ground vegetation under afforested peat is required, which will take account of the gains due to PP of the ground vegetation and roots. Although the ground vegetation was visibly less dense than that of the pristine peatland, the same PP values as used for the pristine peat (Table 10.2) were used for ease of calculation. It is however realised that these values are likely to be higher than in reality (with greater ground vegetation under pristine peat: see Figure 10.3 a and b), and thus will make the forested peatland appear a greater carbon sink than in reality. Reference to Table 10.3 indicates that depending on the rate of PP, the ground vegetation and roots under the forested peat will have an NEE of between 354.91 g $C/m^2/yr$ and 473.91 g $C/m^2/yr$, with an average emission of 406.30 g C/m2/yr. This value for NEE also includes carbon losses from tree root respiration and litter decomposition, and to therefore gain a true value for NEE of the forested peat ground surface, a value for litter addition, and PP of tree roots is required.

In addition to the NEE of ground vegetation and peat, a value for NEE is also required for the trees on the forested peat. As the losses of carbon from tree roots have already been included in the NER of the peat and ground surface, a value for above-ground tree carbon sequestration was required, knowing that the trees at Greenleighton have a yield class of 10 (Forestry commission data). The results of a literature review into above-ground tree carbon sequestration rates are presented in Table 10.3 With a mean value of -235 g C/m^2/yr added to the mean NEE of the forested peat so far calculated, the forested peat would become a much smaller source to the atmosphere of 171.30 g C/m^2/yr. Comparison of this value with a mean loss calculated for the pristine peat of 35.70 g C/m^2/yr, indicates that the forested peat is likely to be a greater source of atmospheric carbon, even when the gains in tree biomass are accounted for. The calculations so far have, however, included losses from tree root respiration on the forested peat, but have not taken account of any gains in carbon from tree root growth nor gains from litterfall. In combination these two values would need to sequester at least 136 g C/m^2/yr to make the forest a smaller source of atmospheric carbon than the pristine peatland. The results from litter collection at Wallington suggest carbon gains from litter of 12 g C/m^2/yr, leaving necessary gains from root growth of 123.3g C/m^2/yr (Table 10.2 and Table 10.3).

	PP (g C/m²/yr)				NER (g C/m²/yr)
Study	AG surface vegetation and roots	AG tree biomass	Leaf litter	BG tree roots	AG surface vegetation and roots, leaf litter, tree roots
Clay (2009)	-181.65				
Rowson (2007)	-127				
Worrall et al (2009)	-246				
Clymo et al (1998)	-189.4				
Garnett (1998)	-229				
Milne and Brown		-125			
Northumberland National Park		-220			
Cannell (1999)		-360			
This study			-12		600.91
Mean carbon flux (gC/m²/yr)	**-194.61**	**-235**	**-12**	**?**	**600.91**
Mean total PP/NER	**-441.61**			**?**	**600.91**

Table 10.3 Afforested peat carbon flux values and calculations using results from this current study and previous studies in the literature, where, AG: above-ground; BG: below-ground; ?: un-measured; where a negative flux represents a sink of carbon and a positive number of source of carbon to the atmosphere

To estimate if these gains are likely, a value for PP of tree roots is needed. Root production rates are however very difficult to measure, and estimates show that in general less than 20% of above-ground production is made up of below-ground production (Waring et al., 1998). The actual carbon content of fine roots is low because they have a low tissue density (Kalyn and Van Rees, 2006), and a study by Kalyn and Van Rees (2006) found no more than 6% of total biomass carbon to be stored in fine roots, and no more than 8% in the coarse roots. A carbon gain in root biomass of 124 g C/m^2/yr is therefore considered unlikely. In addition to the elements of the carbon balance discussed, losses from DOC, dissolved CO_2, and POC following afforestation must also be added. For a quantitative value regarding the impacts on the total GHG balance, values relating to CH_4 are also required. Although no such quantitative values have been found in the literature, this will increase the losses from afforested peatland further still. It is also important to consider that even if the necessary gains in carbon via sequestration in tree roots were possible, this study has assumed thicker ground vegetation under afforested peat than is likely, and for the carbon sequestration due to afforestation to be permanent, it is necessary for all wood harvested to be converted into permanent wood products. The results from this method of assessment, although incomplete, suggest that afforestation of peatlands typical of those in this study will result in an increase in carbon losses to the atmosphere over that of peatlands left in their pristine state.

10.3.5 Further discussion

The results of the work undertaken in this study are conflicting depending upon the method used to estimate the impacts of afforestation on the peatland carbon budget. A major limitation of the quantitative calculations made in Section 10.3.4.1 is the lack of either measured or literature data relating to carbon sequestration in tree root biomass. These results suggest greater carbon losses to the atmosphere under afforested peat than peat in its pristine state, in contradiction to those presented in Section 10.3.4. Although not currently known, a further possible reason for such contradiction relates to the method used in Section 10.3.4, with Equations 10.1 and 10.2 having being proposed as a result of work on a pristine peat catchment and not that of an afforested peatland. These equations indicate carbon fluxes in PP greater in magnitude than those in NER, however until the carbon sequestered into below-ground tree roots can be quantified it is unknown whether the same stoichiometric equation will apply to the afforested peatland studied in this chapter. Other issues responsible for uncertainty in this study relate to confusion over the PP, NEE and NER measurements made in other studies, and their relation to the complete ecosystem, or peatland ground surface.

Despite these limitations, an advantage of this current research over previous studies, is that it contains at least one set of 6 measurements per month from each land-use, and in the majority of cases measurements were made more frequently: approximately once per fortnight. This meant that an average yearly flux could be estimated for both land-uses with confidence, and that calculation did not rely on extrapolation of only a few months' data, and the consequent modelling and prediction of fluxes based on a limited dataset as in Hargreaves et al. (2001). This is especially important in a country such as England, where there is a large seasonal variation in both temperature and water table. To predict annual NER fluxes without accounting for this seasonal variation could result in substantial inaccurate flux values, as NER is often found to show a large amount of seasonal variation (Mielnick and Dougas, 2000).

10.4 Conclusions

Direct measurements made in this study indicate a larger annual NER flux from the ground surface to the atmosphere from afforested peat than from that which has been allowed to remain near to its pristine state. In a similar manner, the total amount of DOC lost from the afforested peatland in a one year period was significantly greater than that from the pristine peat. The greater NER loss from the ground surface under afforested peat was however only significant in October and November, and there were no individual months in which the loss of DOC was significantly greater than that from the pristine peatland. Despite these greater losses in the form of ground surface NER and DOC, a method used to incorporate these results with those from previous literature reveals that the impact on the total carbon and GHG budgets will be positive, suggesting that peatland afforestation in the UK will increase atmospheric carbon sequestration. In contradiction to these results, a method used to produce quantitative values for the impact of afforestation on the peatland carbon budget suggests that peatland afforestation will cause an increase in carbon losses to the atmosphere relative to pristine peat. A lack of data regarding carbon sequestration in below-ground tree roots means though that these results must be taken with caution. Until better methods of measuring the complete carbon and GHG budgets of afforested peatlands can be made, no consensus can be achieved regarding the impacts of afforestation on peatland carbon budgets, and much further research is required. In regard to peatland land-use change at Wallington, no firm land management change suggestions can be made until this necessary further work is undertaken.

Chapter 11

Conclusions

11.1 Introduction

This thesis forms part of a pilot study undertaken by the NT, to identify methods to reduce their carbon emissions, and become a carbon neutral organisation. The focus of the work within this thesis is on the land-based carbon stock of NT land, in particular, SOC, and has investigated the impacts of potential land-use and land-management change which could be made to help the organisation achieve its aim. This thesis does not intend to be a “how to” guide to land carbon sequestration, but instead to add to areas of research on land-use change and carbon sequestration, and to indicate the outcomes of new areas of research which deserve further work.

11.2 A review of the aims and objectives

As indicated in Chapter 1, the objective of the work undertaken in this thesis was to:

- Measure the Wallington Estate’s current land-based carbon stock
- Establish the controls on SOC distribution
- Identify areas/land-uses which are under-saturated in soil carbon, and have the potential for greater storage
- Identify land-use interventions to increase total above and below-ground carbon storage
- Ensure that the results are transferrable to the entire NT Landholding

- Ensure that any land-management change suggestions can be achieved with limited impact on water quality, soil quality and fertiliser use, whilst reducing the reliance on fossil fuel generated products.

The results and findings presented in Section 11.3 will take each objective in turn, and illustrate how each has been achieved, presenting the key findings produced from this research.

11.3 Results and findings

11.3.1 Measuring and estimating the Wallington Estate carbon stock

The initial objective to measure the Wallington Estate's land based carbon stock was achieved by research and soil sampling undertaken and presented in Chapter 2. The results of soil analysis and land-use surveys indicated a total land carbon stock to a depth of 20 cm of 845 Kt C (consisting of 785 Kt in the Estate's soils, and 60 Kt within the above-ground biomass). Nationally available soil and land-use %SOC data measured on soils spanning the UK were also used to estimate SOC distribution at Wallington. Investigation into methods of predicting SOC stock and distribution found that:

- only a small amount of variation in %SOC could be explained by variation in major soil group, using either values specific to the Wallington Estate, or those obtained from national databases
- prediction of SOC stocks using mean %SOC values for major soil groups would not produce an accurate estimate
- Large variations in land-use, altitude and other controls on %SOC beneath individual major soil groups meant that more accurate estimates could be produced if larger scale soil maps were available, and %SOC values specific to individual soil series were applied. The same issues concerning scale were true for estimates based on typical land-use %SOC data.

Estimates of SOC stock made in this thesis were therefore produced using mean %SOC values specific to soil series/land-use combinations. It was concluded that for the best estimates of SOC stock on the Wallington Estate to be made, information relating to the controls on %SOC and their influence would be required. Statistical models developed in this study, could, in the future, be used to predict SOC stocks to the greatest accuracy.

11.3.2 Establishing controls on the SOC stock

Establishment of the controls on SOC was undertaken alongside measurement of SOC stock in Chapter 2, and on the Wimpole Estate in Chapter 5. Analysis of results indicated that:

- Land-management, land-use, land-use history, soil series, soil pH and altitude are significant explanatory variables in %SOC distribution.
- Land-management (the particular farm under which soils are located) can explain the greatest amount of variation in %SOC.

Investigation into land-management techniques via a questionnaire with Estate farmers identified a possible role for basic slag in SOC accumulation under rough pasture; however these findings were not confirmed in a fertiliser trial undertaken in Chapter 4. The greatest %SOCs under farms in arable land-use were located under those which incorporate stubble into their soils following harvest.

11.3.3 Identify areas of under-saturation and potential storage

The results of chapter 1 revealed a large variation in %SOC beneath rough pasture, suggesting that some areas of this land are under-saturated, and if managed correctly could sequester atmospheric carbon. While attempts were made to establish reasons for this variability in Chapters 3 and 4, no firm conclusions have been reached. The extent of %SOC variability in this land-use does though suggest that correct land-management of rough pasture has the potential to greatly increase carbon storage.

With land-use history identified as an explanatory variable in %SOC, areas of improved pasture with a history of arable land-use are likely to be under-saturated, adjusting to equilibrium, and sequestering atmospheric carbon. These areas could be of vital importance to the NT if arable expansion for food production is required in the future, releasing less carbon to the atmosphere than permanent improved pasture upon conversion.

11.3.4 Identify land-use interventions to increase total above and below-ground carbon storage

Attempts to identify land-use interventions to increase total above and below-ground carbon storage were made throughout this thesis. Although many land-use intervention trials indicate a requirement for further research, some strong suggestions though not firm conclusions are apparent.

The results of computer modelling undertaken in Chapter 6 revealed the impact that historic land-use change has had on SOC, indicating that a change into arable and urban land will cause carbon emissions to the atmosphere, but that the extent of the emissions will depend on the current land-use of that which is converted. SOC gains on the other hand would occur with arable conversion to grassland, land-use change out of urban, and land-use change into woodland on mineral soils.

Chapter 7 indicated that incorporation of lump-wood charcoal high in carbon content would sequester large amounts of carbon into land, with no significant increase in NER other than in the first week following charcoal incorporation. Although greater losses of DOC were observed from soils into which charcoal was incorporated, the extent of these losses was small in relation to the amount of carbon contained and sequestered within the charcoal. The results of this trial suggest that charcoal incorporated into arable and forest soils in north east England has the potential to be a land-management carbon sequestration method.

Results from a study assessing the impacts when charcoal is applied as a surface-dressing to pastures (Chapter 9) indicated that land will become a large carbon sink following application. Although greater carbon losses to the atmosphere were observed under land treated with charcoal, these did not outweigh the carbon added. %SOC maintenance under plots treated with 64,000 Kg charcoal/ha suggests that the greater NER observed under charcoal treated plots was not originating from native SOM, but from carbon within the applied charcoal. Although gaseous carbon flux readings indicated reduced PP under land treated with 64,000 Kg charcoal/ha, the appearance of grassland under this treatment in the autumn months was suggestive of improved grass growth.

Research in Chapter 8 observed lower %SOC under SRC willow plantations than arable land, as well as greater NER from the ground-surface. No carbon loss measurements from the above-ground tree or arable crop biomass were however made, nor were any measurements of gains in PP and carbon sequestration under either land-use undertaken. Assessment of such carbon flux pathways and the end use of the products of arable and SRC plantations are required, before any land-management change out of arable land can be encouraged.

Research into the impact of pristine peat afforestation described in Chapter 10 indicates that carbon losses to the atmosphere from soils, ground vegetation and roots will increase following this land-use change. Losses of carbon in the form of DOC were also found to increase with peat afforestation, implying that for peat carbon stocks to be preserved, pristine peatlands should be protected. When the impacts of land-use change on above-ground carbon storage are considered in addition to the carbon stocks of the peatland itself, the impacts on carbon emissions and sequestration become more uncertain. This uncertainty stems from the lack of measurements and results in literature regarding carbon sequestration into tree roots, and the end use of the biomass products. A full life-cycle assessment is required before firm conclusions regarding the impacts of this land-use change on the carbon balance of peatlands can be made.

11.3.5 Ensure the results are transferrable to the entire NT landholding

Soil sampling at Wallington was used to establish the controls on %SOC, in the hope that this would allow an accurate prediction of SOC distribution on other NT estates. In addition to the controls observed at Wallington soil samples taken from the Wimpole Estate in Cambridgeshire identified land-use history as an explanatory variable in SOC distribution, with none of the variation being explained by land-management. With a lack of land-use change on the Wallington Estate in recent decades, and a lack of varying pasture land-management techniques on the Wimpole Estate, this validation study indicated that the use of only one pilot study cannot identify all controls on %SOC, and has thus added to the data from Wallington.

With the most accurate estimate of %SOC distribution at Wimpole being achieved using mean Wallington land-use %SOC values, rather than nationally available %SOC soil or land-use data, it was concluded that the Wallington dataset will be the best to use to predict stocks on other NT landholdings. This validation study has however also indicated that additional controls on %SOC may be identified at other sites, and if time and equipment is available, fieldwork to establish %SOCs at each individual site should be undertaken.

SOC models produced and described in chapter 6 could potentially be used by farmers on each individual land-holding, to assess the impact of any future land-use change decisions they may make. Although the results of all land-use intervention trials in this thesis provide an insight into the effects of land-management change, it cannot be guaranteed that the same results will be true for other areas of the UK until trials are undertaken on varying soil types, and in different environments.

11.3.6 Ensure that any land-management change suggestions can be achieved with limited impact on water quality, soil quality and fertiliser use, whilst reducing the reliance on fossil fuel generated products.

11.3.6.1 DOC

- The application of a variety of fertilisers in Chapter 4 was found to have no effect on soil or run-off water DOC concentration, implying that changes in pasture fertilisation schemes will not result in increased water treatment costs, or be harmful to stream water life.
- The incorporation of lump-wood charcoal into mineral soils caused an increase in leachate DOC concentration, although the increase was small in extent. This was not observed in waters draining organo-mineral soils, or in run-off or soil water from pastures to which charcoal had been applied. The observed losses of DOC in mineral soil leachate must be investigated further before charcoal incorporation advances, to ensure that water quality is not compromised.
- Greater DOC concentrations were observed in soil water draining afforested peatlands than in that from pristine peat. As no firm land-use change suggestion on peat has been reached, these greater losses of DOC under afforested peat, and the impact on stream water and water treatment should be taken into consideration, to help in the decision to afforest or leave peatlands in their pristine state.

11.3.6.2 Nitrate leaching

- No increase or decrease in nitrate leaching relative to the control was observed following the application of any fertiliser in Chapter 4.
- A significant reduction in the nitrate concentration of leachate was observed under mineral soils into which charcoal was incorporated. As these soils had been previously fertilised with ammonium nitrate, it suggests that charcoal incorporation will reduce fertiliser loss from land, bringing economic benefits to landowners, and improvements in the quality of surface waters. Further research is though still required, to ensure that nitrate availability to plants has not been reduced.
- No increase or decrease in nitrate leaching relative to the control was observed in waters draining grassland to which charcoal had been applied as a surface dressing. As this land was unfertilised the consequences of such a land-use intervention on fertilised grasslands are unknown.

11.3.6.3 pH

- Lump-wood charcoal incorporation into soils caused an increase in soil and soil water pH relative to untreated soils. Similar results were observed when charcoal was spread directly onto grassland surfaces.
- Although only small, a difference in pH was also observed in soils and soil water treated with different fertilisers, with a higher pH under plots fertilised with basic slag than nitrogen.
- These results suggest that both lump-wood charcoal and basic slag have the potential to be used as liming agents, and that conversion to use of these fertilisers will both benefit farmers economically, and improve the stream water environment through reductions in water acidification.

11.4 Limitations

As the nature of the work in this thesis is largely field and laboratory orientated, the results must take the limitations associated with such work into consideration. Limitations with regard to land-use intervention trials include:

- The use of a limited number of gas collars, run-off traps, and dipwells, as representative of the land under which they are placed. Although randomised plot designs and sample locations were used in an attempt to overcome such limitations, it is still possible that variations in and between plots prior to land-management intervention could obscure the results of these trials.
- Time and equipment shortages, meaning that a maximum of only six each of run-off traps, dipwells and gas collars could be monitored for each land-use under study. In an ideal situation a greater number of each piece of equipment would be installed, and trials run for longer time periods, allowing the impacts of land-management change over timescales longer than one year to be assessed.
- The issue of uncontrolled experimental variables applies to all land-management intervention trials, and also to interpretation of results in Chapter 3. A limitation specific to research in Chapters 4 and 10 is the uncontrolled variable ‘livestock grazing’.

In relation to the soil sampling campaigns undertaken on the Wallington and Wimpole Estates, limitations in SOC stock estimates are the result of collection of only a limited number of soil bulk density measurements to translate %SOC into SOC stock. This limitation results from time and equipment shortages, but could be overcome by further work.

The use of qualitative methods in Chapter 3 to investigate the role of land-management in SOC accumulation has limitations in the form of the subjective nature of questionnaire responses, meaning that the data was difficult to compile, interpret and analyse.

In relation to the nature of laboratory work, limitations are in existence throughout this thesis. Equipment failure meant that the measurement of anion concentrations in soil and water samples could not be made on all samples, and there was insufficient time available to undertake measurement of sufficient replicates. Although all measurements of %SOC and %OM were made in duplicate, undertaking triplicate measurements would provide more accurate results.

11.5 Recommendations for future work

With rough pasture land-management identified as a major explanatory variable in %SOC variation on the Wallington Estate, there is a requirement to establish the underlying factors within land-management responsible for this variation. The identification in this study of a large spread in %SOC beneath one land-use indicates a great potential for carbon sequestration if all under-saturated rough pasture can be correctly managed. Assessment of the long-term impacts of rough pasture fertilisation schemes and grazing is needed. The outcomes of such research could help not only identify land-use change to sequester carbon, but also to allow more accurate SOC stock mapping, preventing land-use change on areas high in %SOC which may go unidentified if mapped simply by land-use.

- Plot trials where fertilisers are applied annually over a period of several years are required, with continual monitoring of carbon fluxes for the entire trial period, and an assessment of a change in %SOC following many annual applications. To avoid any possible interaction effects grazing animals should be removed from the field, allowing a controlled experiment and the true impact of fertiliser application on grassland %SOC to be established.
- Although the results of Chapter 3 indicated no correlation between rough pasture grazing regime and %SOC, interacting factors and the subjective nature of data collection mean that a controlled experiment is required. Large plot trials should be undertaken where livestock stocking rate is varied between plots, ensuring that all external factors are controlled, with no variations in fertiliser treatment.

The use of lump-wood charcoal as a form of biochar in this study was the result of this material being readily available. If biochar is to be produced from SRC willow and agricultural farm wastes in the future, there is a requirement to investigate whether this material will produce the same results.

- The potential for biochar production from Wallington Estate farm wastes should be established, and methods should be sought to produce this material.
- Plot trials specific to this type of biochar should then be undertaken, under each of the land-uses: arable, forest and pasture.

- Assessment of the impacts on nitrate leaching under fertilised soils is required, to ensure that nitrate availability to plants is not reduced.
- The long-term effect on grassland and arable crop productivity needs to be established.

With the impacts on the complete carbon cycle following land-use change from arable to SRC willow and peatland afforestation beyond the scope of this thesis, the focus of further research concerning these land-use change scenarios should be on:

- Establishing the carbon sequestered in above-ground SRC willow and coniferous tree plantations and roots.
- Undertaking a complete life cycle assessment of each land-use change, as although bio-energy and coniferous forest trees may sequester more carbon into their biomass than arable crops, the end product of each species will be a vital determinant in calculation of the effect of each land-use change on the total carbon balance.

List of appendices

Soil pH

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