National University of Ireland, Cork.

Department of Civil and Environmental Engineering.

Head of Department: Doctor Michael J. Creed.

Supervisor: Professor Gerard Kiely

The Impact of Afforestation on the Carbon Stocks of Irish Soils.

Thesis presented by

Michael Wellock

For the Degree of

Doctor of Philosophy

May, 2011

Table of contents .... Page No.

Declaration

I declare that this thesis has not been previously submitted as an exercise for a degree at the National University of Ireland or any other university and I further declare that the work embodied in it is my own, or else noted.

Michael Wellock

Abstract

Since the 1980s there has been a substantial increase in the forest area of Ireland. Coupled with a projected increase of 1.2 million ha of forest area in the next two decades, afforestation provides Ireland an opportunity to potentially offset substantial carbon dioxide (CO_2) emissions. Afforestation leads to the sequestration of large amounts of C into the forest biomass, however, the effect on soil carbon stocks has been little studied in Ireland. This is crucial if Ireland is to meet it's obligations to the Kyoto protocol.

This study measured the soil organic carbon densities (SOCD) of both afforested mineral and peat soils and the possible factors controlling SOCD change upon afforestation. Amongst all sites there was a mean decrease in SOCD following afforestation. In addition, there was a subsequent increase in the biomass and forest floor carbon density, and when added to the SOCD it resulted in a net gain in ecosystem carbon density following afforestation. The paired site study suggests that planting broadleaf species may be an effective management strategy in promoting SOCD sequestration. However, the chronosequence study found that ash afforestation leads to a decline in SOCD. Additionally, we found that establishing forests on rough grazing (poor grassland) sites or low pH soils may help to minimize SOCD losses.

Few studies have been published on the physical properties of Irish peatlands. We found peat depth to be a key predictor in estimating the SOCD of peatlands. As the SOCD of the high and low level blanket peats are very similar we recommend they be treated as one peat type for future sampling and accounting purposes. The high variance within study sites means there is still uncertainty as to the effects of afforestation on Irish SOCD, and future resampling should be conducted to enhance the findings of this study.

Acknowledgements

My first acknowledgement goes to my supervisor, Professor Ger Kiely, whom I thank for his support and great advice throughout the years of my PhD.

I am also indebted to Christina Megan LaPerle for all the help and hard work that she has put in over the last 3 years whether it be in the field, in the lab or in the office. Thank you to everyone in the lab that has been there throughout my PhD for their help and friendship over the years, with special thanks to Hadrien Robin and Gaelle Treut for their help in the sampling and analysis of the peatlands in the summer of 2009. Also, thank you to Brian Reidy, Tom Bolger, Kevin Black, Brian Tobin, Alistair Pfeifer and all of my other colleagues in University College Dublin and COFORD. Thanks also to all the foresters, landowners and local people that have helped me through the project, and let me sample their land.

I'd like to thank Doctor Creed, head of the Civil and Environmental Engineering, for the use of the facility and Nora O'Sullivan and Claire O'Sullivan for their kind help up in the office.

And finally I'm highly indebted to COFORD- National Council for Forest Research and Development, for the funding of the study Soil Carbon Stock Changes and Greenhouse Gas Flux in Irish Forests (ForestC) and my 3 year PhD fellowship.

1. Introduction

1.1. General introduction

Atmospheric carbon dioxide (CO_2) concentration has increased from pre-industrial values of approximately 280 ppmv to 379 ppmv in 2005 and is increasing at a rate of 1.4 ppmv per year (IPCC, 2007). Much of this increase is due to the release of CO_2 through fossil fuel burning and land use change. Rising atmospheric CO_2 levels are a major driver of climate change and have become a major public concern. The increasing threat of climate change led to the Kyoto protocol to be enacted. As Ireland is a party to the UNFCCC, it is committed to publishing inventories of greenhouse gas (GHG) emissions from sources and removal by sinks, with land use change and forestry being two sectors that require a GHG inventory. Article 3.3 of the Kyoto Protocol allows changes in carbon (C) stocks due to afforestation, reforestation, and deforestation since 1990 to be used to offset inventory emissions.

At the beginning of the twentieth century forests covered only 1% of the total Irish land area (Pilcher and Mac an tSaoir, 1995). Due to the efforts of successive governments there has been rapid afforestation since the 1960s resulting in a 10.0% forest land cover as of 2007 (NFI, 2007a), with 62.8% of forests aged under 20 years (NFI, 2007a). This is largely due to government afforestation grant incentive schemes introduced in the mid 1980s (Byrne and Milne, 2006). This rapid afforestation represents an opportunity for Ireland to offset significant GHG emissions. Under a business as usual scenario, it is estimated that afforestation since 1990 may offset ~22% of the required GHG emissions for the first commitment period, 2008-2012 (Byrne and Milne, 2006).

Reporting on afforestation induced C stock changes must include measurements of change in above- and belowground biomass, litter, deadwood and soil organic carbon (SOC). Biomass and soils are the two biggest C stores within forests, and although it has been well observed that biomass C increases following afforestation (Black et al., 2009), the effects upon soil C stocks are still uncertain (Laganière et al., 2010). Soil C changes following afforestation are influenced by a number of factors, including: tree species, soil cultivation method, pre-afforestation land use, soil properties, stand age, site management, topography, and climate (Guo and Gifford, 2002; Paul et al., 2002; Jandl et al., 2007; Laganière et al., 2010). However, the relative impact of each factor is unclear. Therefore, it is crucial to measure the change in soil C stocks following afforestation in Ireland and simultaneously try and gain an understanding of the controlling factors.

1.2. Aims and objectives

This study investigates the soil carbon stocks of afforested stands within Ireland and attempts to ascertain any possible changes due to afforestation. It consists of a large soil sampling regime across the Republic of Ireland that was undertaken between October 2007 and October 2010.

The study was divided into 3 sub-projects, each with a specific aim:

1. What is the impact of afforestation on the carbon stocks of Irish mineral soils? (Chapter 4).

    a. Assessment of the soil organic carbon (SOC, %), bulk density (g cm^{-3}), soil organic carbon density (SOCD, Mg C ha^{-1}) and forest floor carbon density (Mg C ha^{-1}) of forest and comparative non-forest mineral soils across Ireland.

    b. Investigation of the impact of afforestation on the SOC, bulk density and SOCD of mineral soils, and the factors possibly controlling any change.

2. Soil organic carbon stocks of afforested peatlands in Ireland (Chapter 5).

    a. Assessment of the SOC, bulk density and depth of afforested high-level blanket bogs, low-level blanket bogs and raised bogs across Ireland.

    b. Use of the SOC, bulk density and depth to estimate the soil organic carbon density of Irish peatlands and creation of equations to enable future estimates of afforested peatland soil organic carbon density.

3. Changes in ecosystem carbon stocks in a grassland-ash (*Fraxinus excelsior* L.) afforestation chronosequence in Ireland (Chapter 6).

a. Investigation of the changes in C over time stored within the biomass carbon density (above- and belowground biomass, Mg C ha^{-1}), forest floor carbon density (Mg C ha^{-1}), SOCD (Mg C ha^{-1}) and ecosystem carbon density (Mg C ha^{-1}) of a grassland-ash chronosequence on a brown earth soil.

b. Assessment of the affects of future ash afforestation in Ireland.

1.3. Thesis layout

The thesis contains eight chapters and a list of references. Following the present introduction, Chapter 2 is a literature review addressing the importance of SOC and the impact of afforestation on these stores. The methodologies used to perform this work are presented in Chapter 3. Chapter 4 assesses the impact of afforestation on SOCD of mineral soils and investigates the potential controlling factors. Chapter 5 presents an analysis of the SOCDs of afforested peatlands within Ireland. Chapter 6 investigates the change in ecosystem carbon density following afforestation of grassland with ash (*Fraxinus excelsior* L.). Chapter 7 presents a general discussion of the thesis. Chapter 8 concludes the thesis and provides recommendations for future research which is followed by the reference list in Chapter 9.

2. Literature review

2.1. Climate change and policy

The atmosphere plays a vital role for life on Earth as through the greenhouse effect it maintains the temperature of Earth, creating suitable conditions for life. The greenhouse effect allows short wave energy to reach the Earth while also not allowing long wave radiation to leave. The most important greenhouse gases (GHG); water vapour, carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are found at low concentrations within the atmosphere. Greenhouse gases vary in their effect on the global climate due to their respective atmospheric lifetimes and radiative properties. Each GHG can be expressed as a CO_2-equivalent emission (the emission of a GHG multiplied by its global warming potential) for comparison (IPCC, 2007).

Industrial era human activities, particularly fossil fuel burning and land use change (global deforestation); have led to increases in the concentrations of GHGs in the atmosphere. The carbon dioxide (CO_2) concentration within the atmosphere has increased from pre-industrial values of approximately 280 ppmv to 379 ppmv in 2005 and is increasing at a rate of 1.4 ppmv per year (IPCC, 2007); this drives change in both the local and global climate. In the next few decades Ireland is predicted to see increased temperatures, while also experiencing drier summers and wetter winters, with a greater chance of droughts in the south and east of the country (Ray et al., 2008). It is important that foresters are aware of these potential changes as they will likely impact the productivity of different tree species under different site conditions. Foresters will have to adapt their management strategies for future climate scenarios.

In response to the increasing threat of climate change, the United Nations Framework Convention on Climate Change (UNFCCC) was established with the aim of stabilising GHGs at a level that prevents anthropogenic interference (UNFCCC, 1992). Under the terms of the Kyoto protocol, Ireland is committed to reduce GHG emissions by 13% above the 1990 base year level over the period 2008-2012 (Kyoto Protocol, 1997). Article 3.3 of the protocol requires nations to account and report carbon stock changes associated with land use, land use change and forestry (LULUCF) such as afforestation, deforestation and reforestation. This allows nations to offset CO_2 emissions from afforestation which has taken place since 1990. Consequently, the substantial afforestation that has taken place within the last 20 years provides Ireland with an opportunity to offset a significant amount of CO_2.

2.2. Carbon pools

There are five major global C pools. The largest is the oceanic pool with 38,000 Pg (Pg = 10^{15} g), followed by 5000 Pg stored in the geologic pool (coal, oil and gas), pedologic (soil, 2500 Pg), atmospheric pool (760 Pg) and the biotic pool (560 Pg) (Lal, 2004). These pools are interconnected, and C is constantly cycling between them.

Although the biotic pool is the smallest of the C pools, vegetation is an important C store. However, vegetation is an unstable C store due to quick turnover and man induced land use changes, with global deforestation being one of the main inputs of CO_2 into the atmosphere. Plants take up CO_2 from the atmosphere through photosynthesis where it is stored within the biomass of the plant. Carbon is eventually transferred to the soil when plant material dies and is incorporated into the soil. Plants are the mechanism by which CO_2 is transferred from the atmosphere to the soil (Tomlinson, 2005). Forests contain large amounts of C within the biomass and an estimated 80% of all C stored in the biomass globally (Dixon et al., 1994; Peng et al., 2008; Laganière et al., 2010). Within Ireland, forests contain over 50% of all biomass C on only 10% of land (Cruickshank et al., 2000; Tomlinson, 2006). This proportion will increase in the coming decades as much of the national forest estate was only recently planted, and so have a number of years of growth and potential to sequester greater amounts of C (Tomlinson, 2006).

Approximately 70% of the C stored within the forest ecosystem is stored within forest soil (Dixon et al., 1994; Six et al., 2002a). Soils present a much more stable C store than the forest biomass as the residence time of soil organic carbon (SOC) can be >1000 years within stable fractions (von Lutzow et al., 2006). Three studies have assessed the soil C stocks and their distribution within Ireland. Xu et al. (2011) found that forests contain 10.1% of the national soil C stocks (0-50 cm soil depth), 149 Tg of C with forest occupying 9.1 % of the land area. Eaton et al. (2008) noted similar values; with Irish forests occupying 9.2% of the national land area in 2000 and 10.9% (159.7 Tg of C) of the total national soil C stocks (0-100 cm soil depth). Tomlinson (2005) found that 9.3% of the land cover was forest and comprised 9.4% of the national soil C stocks.

Peatlands are an important store of soil C, with boreal and subarctic peatlands estimated to store between 270-370 Pg of C on only approximately 3% of the Earth's land area (Turunen et al., 2002). This is equivalent to 34-46% of the total carbon (as CO_2) in the atmosphere (IPCC, 2007). Although Ireland contains a small proportion of the world's peatlands, it is a very important C store within Ireland. Peatlands cover approximately 17-20.6% of the Irish land area (Hammond, 1981; Connolly and Holden, 2009), and contain between 53 and 62% of the total C stored in the soils of Ireland (Tomlinson, 2005; Eaton et al. 2008).

2.3. Forms of carbon in the soil

The soil C pool is comprised of 2 main components: soil organic carbon (SOC) and soil inorganic C.

2.3.1. Soil inorganic carbon

The largest amount of soil inorganic carbon (SIC) is found in the form of soil carbonates, such as calcium carbonate ($CaCO_3$) and dolomite ($MgCO_3$) (Batjes, 1996; Schlesinger, 2002), and can account for one-third of the total C in soil (Ming, 2002; Mikhailova and Post, 2006). Carbonates can be formed from the soil parent material (lithogenic) or newly formed as a result of soil processes (pedogenic) (Ming, 2002; Mikhailova and Post, 2006; Wu et al., 2009). Some soil types, such as strongly weathered soils and acidic soils, contain little to no inorganic carbon as the carbonates that were once present in the parent material have been dissolved (Batjes, 1996). High carbonate concentrations are often found in soil formed over calcareous material and soils in dry areas (Batjes, 1996). The carbon stored within the SIC is a more stable C store than SOC, and so is not as susceptible to loss following afforestation, and so studies that measure the change in soil C usually focus on changes in the soil organic carbon (SOC) stores.

2.3.2. Soil organic carbon

Organic matter is formed from a mixture of living biomass (plants, animals, insects, fungi, and bacteria), detritus (recognizable dead biomass) and humus (non-living amorphous flora and fauna residues) with soil organic matter defined as the non-living component of organic matter (Trumbore, 1997; Eaton, 2007). The amount of SOM is controlled by the inputs of organic matter into the soil profile, and losses predominantly due to microbial decomposition but also leaching in dissolved organic carbon and erosion (Trumbore, 1997). As organic matter decomposes over time, it results in the formation of a more complex form of organic matter called humus, this process is called humification. Soil organic matter and humus play an important role in the maintenance of soil properties. Soil organic matter is a major source of nutrients for plant growth as the high cation exchange capacity allows it to become a large reservoir of cationic nutrients. This also adds to the buffering capacity of the soil, and so is less susceptible to pH changes from acids or bases. SOM also improves soil aggregation, which also influences soil aeration, soil temperature, soil strength, root penetration, water infiltration and water retention (Biswas and Mukherjee, 1995).

Soil organic carbon (SOC) is an important component of soil organic matter with SOC making up 58% of SOM (Post et al., 2001). Soil organic carbon is an important store of C globally as it is the largest store in most terrestrial ecosystems (Eswaran et al., 2000; Jobbágy and Jackson, 2000). SOC is conceptually separated into pools representing the lability (ease of oxidation) of SOC, ranging from active/labile fractions (root exudates, microbial biomass and rapidly decomposed components of fresh plant litter that may decompose in a few years to decades) to intermediate fractions with a residence time of decades to a hundred years, to slow/refractory fractions that are stable and may have a turnover time of several thousand years (Trumbore, 1997; von Lutzow et al., 2006). The refractory organic matter pool is responsible for the long term stabilization of C in soils (Falloon and Smith, 2000; von Lutzow et al., 2006). The mechanisms controlling SOC stabilization are often sorted into 3 groups: selective preservation (the stabilization of SOM due to its structural composition), spatial inaccessibility (the location of SOM influences the access of microbes and enzymes for decomposition) and interactions with surfaces and metal ions (the chemical or physiochemical binding between soil minerals such as clay and SOM leads to greater SOM stabilization) (Sollins et al., 1996; Six et al., 2002b; von Lutzow et al., 2006). The mechanisms controlling stabilization of C within the soil are not well understood (von Lutzow et al., 2006). The turnover of SOM and thus SOC depends upon the chemical quality of the C compounds (labile or passive), climate and soil properties such as clay content, soil moisture, pH and nutrient status (Trumbore, 1997; Jandl et al., 2007), and several of these factors can be influenced by afforestation and subsequent forest management (Jandl et al., 2007).

Peatlands are wetland ecosystems with large stores of SOC. This is due to the production of organic matter exceeding the rate of decomposition (Bubier et al., 1995). This imbalance is caused by the inactivity of phenol oxidase under anaerobic conditions which are created by a high water table (Freeman et al., 2001). This leads to an accumulation of chemically labile organic matter (Jandl et al., 2007). However, a lowering of the water table through land management changes (i.e. drainage) can shift the catotelm of the peat from anaerobic to aerobic conditions, promoting the decomposition of SOM (Jandl et al., 2007; Koehler, 2010).

2.4. Soil carbon change following afforestation and controlling factors

It has been well established that the C stored within the biomass increases upon afforestation. However, the impact on soil C stocks and the controlling factors are still uncertain. Afforestation is the establishment of forest plantations that, until then, was not plantation forest, i.e. pastures, cropland, rough grazing, scrub etc. The change in soil C following afforestation is controlled by a number of factors, including: species planted, pre-afforestation land use, soil texture, soil pH, forest age, site management, topography, cultivation method and climate (Thuille et al., 2000; Paul et al., 2002; Jandl et al., 2007; Laganière et al., 2010). There have been a number of studies that have been conducted around the globe assessing the change in soil C following afforestation and the factors that control the change, with differing conclusions. Studies have found no change in SOC stocks after afforestation (Davis et al., 2003; DeGryze et al., 2004; Smal and Olszewska, 2008; Peri et al., 2010), others found an increase in SOC stocks (Post and Kwon, 2000; Guo and Gifford, 2002; Hooker and Compton, 2003; Morris et al., 2007; Black et al., 2009; Mao et al., 2010) while some have found a decrease in SOC stocks (Parfitt et al., 1997; Alberti et al., 2008). However, many studies have shown the same trend, a decline in SOC stocks after afforestation, before reversing to increase with age potentially matching pre-afforestation levels and possibly surpassing them (Romanyà et al., 2000; Paul et al., 2002; Turner et al., 2005; Zerva et al., 2005; Huang et al., 2007; Hu et al., 2008; Laganière et al., 2010; Mao et al., 2010).

Soil cultivation methods are employed so as to provide more suitable soil conditions for the establishment of trees, and have the potential to significantly influence the future development of a stand (NFI, 2007b). Soil cultivation methods can improve planting conditions in a number of ways; competition reduction, improved water infiltration, better drainage and reduction of soil strength so as to allow better root penetration (Turner and Lambert, 2000). The soil disturbance also aims to release nutrients for uptake by the trees by enhancing decomposition of SOM through changing the microclimate (Johansson, 1994). A number of different cultivation methods are employed in Ireland for site preparation and they differ in the level of disturbance to the soil. They range from a low disturbance method such as pit planting (use of hand digging with no mechanical help), to medium disturbance methods such as mounding and agricultural ploughing (mechanical site preparations which creates drains or holes, respectively, but leave large areas undisturbed), and to high disturbance cultivation methods single mould ploughing (SMB) and double mould board ploughing (DMB) (use of heavy machinery to prepare large areas of the forested area) (NFI, 2007b). Soil disturbance during site preparation is designed to promote the decomposition of organic matter so as to release nutrients for uptake into the trees, but it leads to an increased loss of C from the soil. Johnson (1992) in a review of the effects of soil cultivation on soil C stocks noted a net loss of soil C due to disturbance, with soil C loss increasing with the intensity of soil disturbance (Örlander et al., 1996; Schmidt et al., 1996; Mallik and Hu, 1997; Bock and Van Rees, 2002; Laganière et al., 2010). Soil disturbance can stimulate soil C loss through a number of mechanisms; breaking soil aggregates that can increase the availability of SOM to decomposition; can stimulate microbial activity through aeration and modification of soil surface environment (temperature and water content); and through the mixing of fresh organic material into the soil. (Örlander et al., 1996; Nouvellon et al., 2008). However, site cultivation also promotes tree growth through increased nutrient availability, water infiltration and root development (Jandl et al., 2007; Nouvellon et al., 2008). Over time this may lead to an increase in the organic matter input to the soil from the larger NPP of the forest. It has been suggested that the C sequestered into the larger tree biomass may outweigh the soil C lost from the soil due to site cultivation (Jandl et al., 2007).

There is a decline in the input of organic matter from the trees to the soil following afforestation. Young trees have a low net primary productivity (NPP) and so little organic material enters the soil via the roots or litterfall. The decline in litter input following afforestation varies, depending upon the NPP of the vegetation prior to afforestation and that of the forest species planted.

As the forest ages, a number of factors lead to the soil switching from losing C to beginning to gain C. The NPP of the forest increases with age, and over time the organic matter input into the soil becomes greater than the decomposition of SOM and so the soils switch from losing C to begin to sequester C (Paul et al., 2002; Turner et al., 2005). The soil microclimate also changes as the forest ages. As the forest canopy closes, the soil surface temperature and moisture content are lower than the soils of grasslands and croplands due to shading and greater transpiration rate, which may lead to a reduction in SOM decomposition rates (Paul et al., 2002; Laganière et al., 2010).

Within forest soils it has been shown that there is an enhanced protection of SOC. Del Galdo et al. (2003) found that following afforestation there was a greater stabilization of C that was present in the soil prior to afforestation into more microaggregates (53–250 μm) and silt & clay (<53 μm) and Six et al. (2002a) noted a greater amount of soil C in microaggregates and microaggregates occluded within macroaggregates (250–2000 μm). Not only was there an increase in SOC following afforestation, but it also had a longer residence time.

The rate at which SOC will decline or increase following afforestation is controlled by a number of other factors, along with NPP and soil disturbance, which will be discussed below.

2.4.1. Pre-afforestation land use

The pre-afforestation land use of a forest has a large effect upon SOC stocks following afforestation (Paul et al., 2002). Laganière et al. (2010) reports that the previous land use explains much of the variability in soil C change following afforestation of agricultural soils. In Ireland a large proportion of recent afforestation was upon pasture and rough grazing land, this is because farmers were encouraged to plant forest upon their land by grant incentives from the government. Studies have shown that pastures and grasslands can store an equal amount or more soil C than that of forests (Corre et al., 1999; Franzluebbers et al., 2000; Turner and Lambert, 2000; Garten Jr and Ashwood, 2002; Laclau, 2003). Guo and Gifford (2002) noted a loss of 10% of SOC from conversion of pasture sites to plantation forest, which is also seen in Paul et al. (2002). However, Laganière et al. (2010) recorded a small increase of 3% in pasture sites and 10% in grasslands. There are a number of reasons that pasture sites can lose C following afforestation. Herbaceous plants and trees have different biomass allocation strategies that affect the amount of SOM that enters the soil in each ecosystem. In trees, a large proportion of the biomass is concentrated in the trunk in the aboveground biomass, while herbaceous plants allocate most of their biomass into the belowground biomass of the root system (Cerri et al., 1991; Kuzyakov and Domanski, 2000; Bolinder et al., 2002). The tree root system lives longer than that of herbaceous plants, and so the annual turnover of organic matter into the soil from dying grass roots are larger than that from the trees (Kuzyakov and Domanski, 2000; Guo and Gifford, 2002; Guo et al., 2007). These two factors lead to the roots of the pasture and rough grazing sites having a greater SOM input to the soil than the forest sites. Guo et al. (2007) reported a 20% reduction in soil C stocks following afforestation of a native pasture (dominated by *Themeda triandra*) with *Pinus radiata* in Australia and found that it was in at least part due to a 36% greater mass of fine roots in the pasture that also had a shorter turnover time than those in the forest. The roots of the grass provided 3.6 Mg C ha^{-1} $year^{-1}$ to the soil annually, larger than the 2.7 Mg C ha^{-1} $year^{-1}$ added by the *Pinus radiata* roots. The productivity of the pre-afforestation land use can also affect the change in soil C stocks following afforestation. Those sites with higher productivity lead to larger inputs of SOM into the soil and thus C sequestered into the soil. This level of sequestration cannot be matched by young trees and so it leads to a decrease in soil C. Sites with a lower productivity will see a smaller decline in SOM inputs into the soil following afforestation (Lima et al., 2006; Scott et al., 2006). The soils of pasture and rough grazing sites are often left undisturbed in Ireland, except for trampling of the grazing cattle, sheep and horses which allows the soil to accumulate C without any large losses. It has also been recorded that scrub sites lose soil C following afforestation. Guo and Gifford (2002) recorded a 13% loss of soil C following afforestation of non-plantation forest sites. This is likely due to the increased disturbance of the soil through removal of the trees from the scrub site (deforestation) and subsequent afforestation. Deforestation of a site can cause large soil disturbance and lead to a loss in soil C (Guo and Gifford, 2002).

2.4.2. Climate

Climate plays a large role in the carbon stocks of soils. It has been shown in the literature that soil C change following afforestation varies between climate zones. Laganière et al. (2010) noted that the boreal zone sites decreased in soil C following afforestation, the SOC of temperate continental, subtropical and tropical soils increased following afforestation, with the temperate maritime zone showing the highest gains. The climate zones are based on Köppen's classification (McKnight and Hess, 2007; Laganière et al., 2010). The soils of boreal zones have the greatest ability to store SOC as the low temperatures suppress decomposition, which is why the majority of peatlands are found in the boreal zone. However, the low temperature also inhibits tree growth, leading to a low input of SOM into the soil (Ritter, 2007), possibly explaining the loss of soil C seen in Laganière et al. (2010). Boreal forest soils have the greatest potential to sequester C, but it may take a long time for it to happen (Ritter, 2007). The high precipitation and temperature of the tropical regions stimulate tree growth, leading to a higher organic matter input into the soil, however, these same conditions also increase SOM decomposition and so limit the capability of tropical soils to sequester C (Paul et al., 2002; Lal, 2005). There is a higher NPP in temperate forests than in the boreal zones due to the warmer temperatures, but not so high as to promote rapid SOM decomposition, and so have the greatest ability to sequester C quickly into the soil following afforestation.

Although there are higher SOC levels in high rainfall areas, it has been seen that sites with higher rainfall lose more SOC than drier sites (Guo and Gifford, 2002; Jackson et al., 2002; Kirschbaum et al., 2008). Guo and Gifford (2002) found that in afforestation of grasslands there was little change for sites with an annual precipitation <1200 mm, and a significant loss of soil C of 10-15% and 23% for sites with a precipitation of 1200-1500 mm and >1500 mm, respectively. Guo and Gifford (2002) also noted that conversion of non-plantation forest to plantation forest showed little change for sites with precipitation <1500 mm, while sites >1500 mm recorded a loss of 56% from pre-afforestation levels. Kirschbaum et al. (2008), using the model CenW found that in areas of high rainfall, larger amounts of nitrogen were leached from the soil, leading to a decline in the soil C stocks, while the drier sites had little leaching of C and so did not lose as much C from the soil. However, in China, Shi and Cui (2010) noted that all sites saw an increase in soil C following afforestation despite the precipitation level. This highlights the need for more detailed examination of precipitation levels in soil C change following afforestation.

2.4.3. Soil texture

The soil texture of a site can have a significant effect on the ability of a soil to sequester C following afforestation; with fine textured soils (silt and clay) having a greater capacity to store C. Laganière et al. (2010) recorded a significant increase in the amount of C sequestered into the soil following afforestation with an increasing clay content. This result was also seen in Paul et al. (2002). The adsorption of SOM on the mineral surface of clay and silt creates stable organomineral complexes, which helps to stabilises the SOM from decomposition (von Lutzow et al., 2006). The large surface areas of silt and clay (especially clay) provide a much greater reactive surface to form complexes with SOM (Sollins et al., 1996; Hassink, 1997; Torn et al., 1997; Six et al., 2002b). The clay type can also impact the amount of C a soil can sequester. Clay minerals with a higher cation exchange capacity (CEC) and large surface area such as vermiculite have a greater potential to bind with SOM than clays with a lower CEC and smaller surface area (Six et al., 2002b).

2.4.4. Soil pH

The soil pH can affect both the input of SOM in the soil, and also affect SOC accumulation following afforestation. Low pH can inhibit the activities of soil micro-organisms that decompose SOM and so can lead to the preservation of SOM inputs into the soil (Motavalli et al., 1995; van Bergen et al., 1998; Andersson and Nilsson, 2001; Aciego Pietri and Brookes, 2008). However, Laganière et al. (2010) found the opposite trend with an increasing SOC accumulation following afforestation with increasing pH (<5 pH = 10%, 5-7 pH = 13%, >7 pH = 22%). However, Laganière et al. (2010) also found that the soil pH had no significant effect on soil C stock changes. Laganière et al. (2010) suggests that the SOC increase with pH may be due to low pH soils retarding tree growth due to nutrient deficiencies, leading to a decline in the inputs of SOM into the soil from trees compared to soils with a higher pH (Augusto et al., 2002; Laganière et al., 2010). This highlights the need for more research to fully distinguish the effect pH on soil C stocks following afforestation.

2.4.5. Tree species

Choice of tree species planted can have a large impact upon the change in soil C stocks following afforestation (Guo and Gifford, 2002; Resh et al., 2002; Vesterdal et al., 2002; Paul et al., 2003; Lemma et al., 2006; Laganière et al., 2010; Shi and Cui, 2010). Laganière et al. (2010) found that the species planted plays a major role in the recovery of soil C, the second strongest predictor behind previous land use. Broadleaf species sequestered significantly more C (25% increase on pre-afforestation levels) than pine forests and conifer forests when planted on agricultural land, which sequestered 12% and 2%, respectively. Shi and Cui (2010) in a review of the effects of afforestation on soil C stocks in China, found that the average change in soil C in 0-40 cm of broadleaf (5.92 $gCm^{-2}\ yr^{-1}$) and mixed forests (4.88 $gCm^{-2}\ yr^{-1}$) were significantly larger than conifer forests (-0.90 gCm^{-2} yr^{-1}), which lost C over time. Guo and Gifford (2002) in a meta-analysis of soil C under different land use conversions, found that afforestation with conifers of both pasture and non-plantation forest sites saw a significant loss of soil C, while the broadleaf forests saw a small loss. Broadleaf forests can sequester a greater amount of C into the soil as they have a larger and deeper root system (Strong and La Roi, 1983) than conifer species, which are shallow rooting (Jandl et al., 2007). The higher belowground biomass of the broadleaf forests produce a larger SOM input into the soil than that of conifer species, leading to a greater effect on soil C. The broadleaf species may also have a greater ability to sequester C in deeper soil horizons due to the larger root biomass, which is important, as the turnover time and chemical recalcitrance of SOM increases with depth of soil (Lorenz and Lal, 2005).

Tree species also affects the amount of C stored in the forest floor, which accumulates at quicker rates than that of the soil (Vesterdal et al., 2002; Vesterdal et al., 2008). A number of studies have shown that afforestation with conifer species has led to a decrease in soil C stocks (Guo and Gifford, 2002; Markewitz et al., 2002; Guo et al., 2007; Laganière et al., 2010). However, conifers sequester a greater amount of C into the forest floor than broadleaves (Vesterdal and Raulund-Rasmussen, 1998; Vesterdal et al., 2008) due to the slower decomposition of the conifer litter. It is important to assess the amount of C stored within the forest floor as studies have found that when the forest floor C stocks are added to the soil C stocks the conifer species switch from losing C to gaining C following afforestation (Paul et al., 2002; Laganière et al., 2010) .

2.4.6. Plantation density

It is believed that the plantation density of a forest stand affects soil C stocks following afforestation. Tree planting density will impact the soil microclimatic conditions (temperature, moisture) as the higher density stands are likely to have a larger canopy cover, shading the soil and lowering its temperature, and thereby suppressing SOM decomposition (Laganière et al., 2010). The higher density of trees means that the total forest biomass is larger than low density stands, and so leads to a larger litter input of organic matter into the soil from those stands. It is due to this that Turner et al. (2005) recommend increasing the initial plantation density to promote C sequestration into the soil. However, recent work has shown that the plantation density has no impact on the soil C stocks. Laganière et al. (2010) found that both high and low density plantations increased SOC stocks after afforestation, and although Davis et al. (2007) found a greater root density within the higher density stands; there was no change in the soil C between the 3 stocking density sites. It is unclear as to why the literature shows no change in soil C between different tree densities, and so more work must be done to assess why higher density stands are not sequestering more soil C as would be expected.

2.4.7. Fertiliser and nitrogen fixers

The use of fertilisers or the planting of nitrogen (N) fixing trees can increase the C stored within the soil. Fertilisation of nutrient poor sites can lead to an increase in the NPP of forests as they may have previously been N-limited (Nohrstedt, 2001; Hoegberg et al., 2006). This leads to an increase in organic matter input into the soil through rhizodeposition and litterfall. Studies have also shown that fertilisation leads to a decrease in the C/N ratio of the litter material, which decomposes quicker than without fertilisers (Parker et al., 2001). Although there is an increase in the decomposition of fresh material, the increased N can also reduce decomposition of old litter and recalcitrant SOM which could affect the long term store of soil C (Jandl et al., 2007). The decreased decomposition of recalcitrant SOM has been observed under both fertilisation experiments and in planting of N-fixing trees. Fertiliser and N-fixing species induced increases in soil C may be due to a number of reasons. One possibility is increased decomposer efficiency due to a shift in community to organisms that are more efficient decomposers, but that have a larger N requirement such as the suppression of ligninolytic decomposers for increased cellulose decomposers. A decrease in the growth rate of decomposers and an increase in the rate of recalcitrant material formation has also been suggested as possible reasons for the increase in soil C (Ågren et al., 2001; Resh et al., 2002; Franklin et al., 2003; Jandl et al., 2007).

2.4.8. Peatland afforestation

Prior to planting, drains are installed within peatlands to lower the watertable and improve growing conditions. However, lowering of the watertable, from both the drainage and also increased evapotranspiration from the forest as it grows, can impact the carbon stocks of a peatland. The lowering of the watertable exposes the usually anoxic catotelm to oxygen. This leads to the growth of microbial aerobic decomposers (Chmielewski, 1991; Byrne and Farrell,

2005) which enhance the rate of organic matter decomposition and loss of carbon as CO_2 (Lieffers, 1988; Bridgham et al., 1991) and leads to a decrease in carbon stocks (Braekke and Finér, 1991; Sakovets and Germanova, 1992). However, as the forest ages there is an increase in the vegetative input of carbon to the peat through roots, litterfall and forest harvest residues (Anderson et al., 1992; Minkkinen and Laine, 1998; Reynolds, 2007) which can increase peat C storage. These uncertainties highlight the need to sample peat properties to assess the current carbon stocks of afforested peatlands.

2.5. Methodologies used to detect soil carbon change

Three types of study designs are used to measure changes in soil C following afforestation: paired sites, chronosequence studies and retrospective/re-sampling design (Turner and Lambert, 2000; Laganière et al., 2010). The paired site method uses a comparison between a forest site and an adjacent non-forest control site. The control site is selected so that all site characteristics are the same i.e. soil type, elevation, slope, etc. The only difference between the sites is the current land use. The land use of the non-forest site is the same as that of the forest site prior to afforestation. The paired site method allows for the comparison of several variables (tree species, soil properties, planting density, etc.) and abiotic factors (climate, topography, etc.) (Resh et al., 2002; Zinn et al., 2002; Del Galdo et al., 2003; DeGryze et al., 2004; Pinno and Bélanger, 2008) and is also simple and inexpensive to develop (Laganière et al., 2010). The limitations of this method are that it only measures one point in time and that there are uncertainties in the site characteristics that are assumed to be similar amongst sites (forest site and non-forest) (Laganière et al., 2010).

The chronosequence method is an extension of the paired site method, and uses a combination of paired forest sites with the same characteristics, differing only in age, so as to measure changes in soil C over the age of the forest (Zou and Bashkin, 1998; Markewitz et al., 2002; Farley et al., 2004; Wang et al., 2006; Black et al., 2009; Mao et al., 2010). A chronosequence can also have a control site as in the paired site methodology. However, like the paired site methodology, the chronosequence method is limited, in that the similarity in site characteristics is assumed. The validity of a number of chronosequence studies have been called into question as the basic premise of these chronosequences had not been validated (Johnson and Miyanishi, 2008). They suggest that chronosequence and paired site studies should attempt to validate the premise of these sites (i.e. all sites have the same pre-afforestation land use) prior to sampling.

Retrospective/resampling design re-samples the same soil over a period of time (Binkley and Resh, 1999; Jug et al., 1999; Richter et al., 1999). This is the least biased methodology as it removes the error associated with differences between sites (Laganière et al., 2010). The one limitation of the resampling design is that it measures changes in real time, and so it is difficult to assess afforestation induced changes as forest rotation lengths in Ireland can be 40 to 100+ years. Laganière et al. (2010) noted in a meta-analysis of soil C changes following afforestation of agricultural land, that the paired site methodology was the most used method (65%) and the soil C of the sites measured under the paired plot methodology increased by 8.8% on pre-afforestation values, the chronosequence sites increased by 2%, and the retrospective design sites decreased by 3.6%. Laganière et al. (2010) suggests that the paired site methodology may be overestimating the change in soil C by 12.4%. They state that this possible error may be due to the sites not being validated for similar characteristics (soil type, elevation etc.), but the exact reason was not well understood.

These methods can also be used to measure changes of other soil properties (soil N, P, pH, etc) (Piccolo et al., 1994; Veldkamp et al., 1999; White et al., 2004; Banning et al., 2008; Uselman et al., 2009), changes in soil fauna (Insam and Domsch, 1988; Allison et al., 2005; Shi et al., 2006), ecosystem properties (Pare and Bergeron, 1995; Bond-Lamberty et al., 2004; Kashian et al., 2005; Uselman et al., 2007) and can measure land use changes other than afforestation (Baer et al., 2002; Breuer et al., 2006; Zavaleta and Kettley, 2006; Ostertag et al., 2008).

The variance of soil C stocks impacts the ability of a study to measure a statistically significant change in soil C following afforestation. The variance of soil C stocks can be affected by a number of factors that must be considered when designing the sampling methodology. Variance and mean carbon content have been shown to be significantly and positively related (Conen et al. 2005). Conen et al (2005) show that due to higher variance with increasing SOC concentration, the number of samples needed to detect a significant change can vary from 10 to several thousands. This is especially a problem for peat soils in Ireland and so care must be taken in development of the soil sampling protocols of peats. Soil disturbance (whether it is due to pre-planting disturbance, windthrow, burning of harvesting material), spatial scale of sampling (detection of changes over local, regional, national etc) time-scale of sampling (changes in SOCD following afforestation can be seen within a few years, whereas effects of climate change can take decades) can also affect the variance of soil carbon stocks (Conen et al. 2005). The variance of soil carbon stocks limits the ability of the paired site, chronosequence or resampling methods to detect changes in soil C and so consideration of the number of samples to be taken, the number of sites to be measured or the age of forest are of great importance in sampling strategy design. However, these decisions are often decided upon an ad hoc basis (e.g. once every six months) or determined by economic or time constraints (Smith et al., 2002; Smith, 2004). To help determine the number of samples, number of sites and how long it will take to detect changes, papers have been published to help improve studies by providing the sampling intensity required to measure soil C change at both the local and national level (Garten and Wullschleger, 1999; Smith, 2004; Saby et al., 2008).

The most commonly used method in determining soil C stocks is the volume-based method which multiplies the SOC concentration (%) by bulk density (g cm^{-3}) to a fixed depth. When assessing the effects of land use or management changes on soil C, the volume-based method assumes that bulk density is static through time (Markewitz et al., 2002; Sartori et al., 2007; Mao et al., 2010). However, soil bulk density varies spatially and temporally (Amador et al., 2000; Gifford and Roderick, 2003; Kulmatiski and Beard, 2004; Lee et al., 2009; Wuest, 2009). The volume-based method does not account for variations in soil mass and this introduces uncertainty in SOC stock changes (Ellert and Bettany, 1995; Markewitz et al., 2002; Murty et al., 2002; Goidts et al., 2009; Lee et al., 2009; Mao et al., 2010). The equivalent soil mass (ESM) correction was proposed by Ellert and Bettany (1995) as a more reliable method to determine changes in soil C stock changes and a number of studies have used it since (Ellert and Bettany, 1995; Carter et al., 1998; Yang and Wander, 1999; Gifford and Roderick, 2003; VandenBygaart, 2006; VandenBygaart and Angers, 2006; Lee et al., 2009; Wuest, 2009; Mao et al., 2010). The ESM is defined as the reference soil mass per unit area chosen in a layer and the equivalent C mass (ECM) is the C stored in an ESM (Ellert et al., 2001; Mao et al., 2010).

2.6. National forest inventory (NFI)

In November 2004, the National Forest Inventory (NFI) began collecting data on the current extent and state of Irish forests (NFI, 2007c). The NFI sampled forest plots across the country that were selected from a randomised systematic grid sample design. After a pilot study in Co. Wexford, a grid density of 2 x 2 km grid was placed over the total land base of Ireland (6,976,100 ha). Aerial photos were used to determine whether the land use was forest or non-forest, and areas that were found to be forest became the focus of the field survey. Permanent inventory plots were established at random locations within 100 m of the 2 km by 2 km grid intersections (NFI, 2007c). Data on the forest was collected within each plot, a circle with a radius of 12.62 m. A total of 1,742 forest plots were measured for a number of forest characteristics, such as; forest management, including forest ownership and forest type; forest structure; forest diversity including species composition; production; damage; deadwood; regeneration, and site characteristics including altitude, slope and soil type (NFI, 2007b). Data collection for the NFI was completed in November, 2006 and the data was published in June, 2007 (NFI, 2007a; 2007c).

3. Materials and methods.

3.1. Paired plot methodology

3.1.1. Site selection

The sites for the paired site study (Chapter 4) were selected from the NFI database (NFI, 2007a). The NFI database includes a large amount of sites that are unsuitable for this study and so the 1,742 forest plots were filtered for suitability. The database was screened so that sites that were inaccessible (those sites that were physically inaccessible by foot) and non-afforested were removed. Reforested and non-plantation forestry sites were not sampled because the focus of this study is afforested sites, as this represents the major land use change in the last few decades and for decades to come (Department of Agriculture, 1996). The database was further screened for age so that those sites that were under 15 years of age were removed. One of the initial aims of the study was to attempt to find the point where the soil organic carbon density (SOCD) matches pre-afforestation levels and begin to increase upon them. Fifteen years was chosen in consultation with other scientists as studies had shown that the soil C of forest sites reached pre-afforestation levels at around 20 years (Davis and Condron, 2002). The database was also screened to exclude soil types that represent less than 1% of the total forest area (fen peat, regosols, lithosols and rendzinas) as they could not provide a suitable sampling size.

From the screened database the sites were categorized by soil type (brown earth, gley, brown podzolic and podzol) and forest type (conifer, broadleaf and mixed). Conifer and broadleaf forests were defined as such when the dominant forest type represented greater than 81% of the canopy cover, while the mixed forests were a combination of both conifer and broadleaf species where the smallest one makes up at least 20% of the canopy cover (NFI, 2007b). It was from these groupings that the final sites were selected. If a grouping had less than 6 sites it was rejected as the small number of sites did not allow for an appropriate sample size. From these groups, 6 sites were randomly selected for sampling, the remaining sites were set aside as backups if necessary. It was at this point that the sites were split between this study, ForestC and a companion project, CARBiFOR II, based in University College Dublin. The sites that were in the northern counties of Ireland (Dublin, Sligo, Donegal etc) were sampled by the CARBiFOR II project team and the sites in the southern counties (Cork, Tipperary, Kerry etc.) were sampled by the ForestC project team. Each group sampled 21 sites, with the ForestC sites presented in table 4.1.

The paired site method was used to determine the impact of afforestation on soil organic carbon density and a number of techniques were used to try and identify the pre-afforestation land use. The first was to interview the forest manager or owner. This was the most effective method as some were involved in the planting of the site, especially private owners, and so had extensive knowledge, or was in possession of records of the land use prior to planting. However, this was not always satisfactory, and other methods were used. An additional technique was to interview local people in the area to identify local knowledge regarding the previous land use. Another technique was to use maps from 1900 to identify the previous land use. The 1900 maps were not used as stand-alone evidence but used to back up information from the other methods of inquiry described above.

Once the pre-afforestation land use had been determined, a suitable site was selected from the local area for sampling. The non-forested site was selected so that it had the same land use as the forest prior to afforestation, the same soil type, slope, an elevation within 20 meters (Eaton, J, personal communication), and a distance within 1.5 kilometres of the forest. Not all selected forest sites were suitable for sampling for a number of reasons. The most common problems were that the sites had been harvested since they were sampled by the NFI or there may not have been an appropriate site in the area to represent the pre-afforestation land use. These sites were removed from the database and were replaced by randomly chosen sites from the list of backup sites.

3.1.2. Field sampling

Sampling of the paired plot sites took place between October 2007 and May 2009, and the sampling design was adapted from Davis et al. (2004). At both the forest and non-forest sites a 20 m x 20 m square was setup and divided into four, 10 m x 10 m quadrants. The soil at each site was sampled to a depth of 30 cm. Within each quadrant at a preselected random point a soil pit was dug for soil classification and bulk density sampling. Bulk density samples were taken at depths of 0-5, 5-10, 15-20 and 25-30 cm using stainless steel bulk density rings (Eijkenkamp Agrisearch Equipment BV, Netherlands) of 8 cm diameter by 5 cm height. Around the pit, 8 holes were sampled, to be analyzed for soil organic carbon (SOC, %), using a soil auger (Eijkenkamp Agrisearch Equipment BV, Netherlands), to the depths of 0-10, 10-20 and 20-30 cm. The forest floor was sampled at the forest sites and the scrub non-forest sites, however, the organic layer of the rough grazing and pasture non-forest sites were not sampled. At three points within each quadrant, the forest floor was sampled from 0.1 m^2 square plots. Each forest floor sample was separated into 3 separate samples, fine woody debris (material with a diameter >2.5cm and <7.5 cm), litter (non decomposed material of diameter <2.5 cm), and F/H layer (decomposed material that has not mixed with the soil).

To summarise, at each of the 21 sites we obtained for both the forest and paired site: (1) 32 samples for SOC at depths of 0-10, 10-20 and 20-30 cm using the soil auger: (2) 4 samples of bulk density at depths 0-5, 5-10, 15-20 and 25-30 cm using the stainless steel rings: (3) and 12 samples of the forest floor. The distribution of samples within the 20 m by 20 m plot can be seen in Figure 3.1.

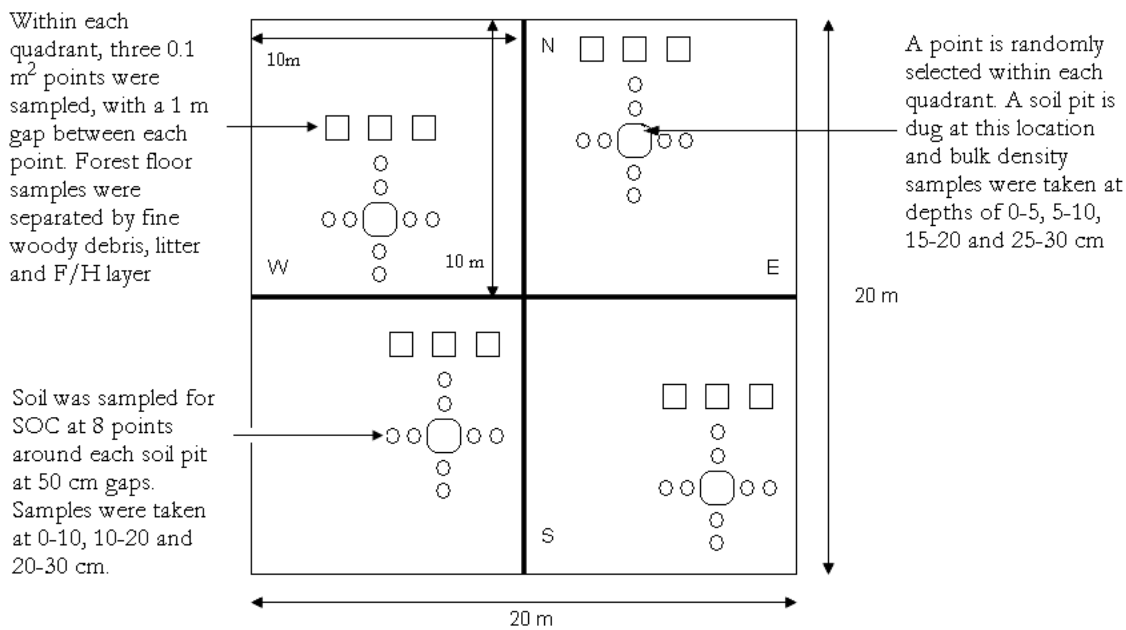

Figure 3.1. The sampling distribution within the 20 m x 20 m plot used at each forest and non-forest site.

3.1.3. Sample analysis

All soil and forest floor samples were stored at 4 ^{o}C before being dried at 55 ^{o}C (a low temperature was used so as to prevent the combustion of organic matter from the soil samples) until a constant dry weight was achieved. The augured soil samples were sieved to <2 mm (the 8 soil samples were bulked by depth within each quadrant by equal volume). All soil samples were tested for the presence of carbonates. Any samples that tested positive for carbonates were treated with sulphurous acid to remove the carbonates (Nelson and Sommers 1996). The soils of sites CBE1, CG4, MBE1 and MBE2 were treated for carbonates. The forest floor samples were bulked for each forest floor type within each quadrant by equal volume. The bulked soil samples and bulked forest floor samples were ground to a fine powder before combustion in a C/N analyser (Elementar - Vario Max CN) to determine the C content (%).

The texture of the mineral soil was analysed within the paired plot study within 2 randomly chosen quadrants for each depth using the hydrometer method, ASTM D422, 2002 and measured for pH using 1:1 soil to water using a LabFit AS-3000 pH analyser (McLean, 1982).

The bulk density samples were sieved to 2 mm and were calculated using equation (3.1):

$$\rho_d = \frac{MA}{SV - CFV} \qquad (3.1)$$

where ρ_d = bulk density (g cm^{-3}); MA = mass of dry sample <2 mm (g); SV = sampler volume (cm^3); CFV = >2 mm coarse fraction volume (cm^3)

The bulk density samples were converted to 10 cm depths to match the SOC data. The soil mass was calculated using equation (3.2).

$$M_i = \rho_{di} \times h \times 10^4 \qquad (3.2)$$

where M_i = dry soil mass (Mg ha^{-1}) at the i^{th} depth (i =1,2,3 corresponding to the 0-10, 10-20 and 20-30cm depths); h = depth of soil layer (m); 10^4 = unit conversion factor (m^2 ha^{-1}).

The soil organic carbon density of the soil to a fixed depth (volume-based method) was calculated using equation (3.3).

$$C_{i,vol} = conc_i \times M_i \quad (3.3)$$

where $C_{i,\mathrm{vol}}$ = soil organic carbon density, volume based method (Mg C ha^{-1}); $conc_i$ = SOC (%)

The equivalent soil mass method uses the soil mass of each soil layer sampled at the non-forest site as the ESM for the comparative forest soil layer, using equation (3.4), the equivalent carbon mass can be calculated using equation (3.5) (Lee et al., 2009).

$$M_{i,add} = M_{i,equiv} - M_i \quad (3.4)$$

$$C_{i,esm} = C_{i,vol} - conc_{i+1} \times M_{i-1,add} + conc_{i-1} \times (M_{i,add} - M_{i-1,add}) \quad (3.5)$$

where $C_{i,\mathrm{esm}}$ = the soil organic carbon density, equivalent soil mass method (SOCDesm, Mg C ha^{-1}); $M_{i,\mathrm{equiv}}$ = non-forest soil mass (Mg ha^{-1}); $M_{i-1,\mathrm{add}}$ and $M_{i,\mathrm{add}}$ = the additional soil masses that are used to estimate the ESM (Mg ha^{-1}); $conc_{i-1}$ and $conc_{i-1}$ = SOC concentration (%) for the additional soil mass determined by the location of soil mass used for the corrections between the soil layers at each sampling time. For more information on the ESM method see Ellert and Bettany (1995), Ellert et al. (2001) and Lee et al. (2009).

The soil organic carbon density (0-30 cm) of the forest sites were calculated using equation (3.6) and the soil organic carbon density (0-30 cm) of the non-forests sites were calculated using (3.7).

$$C_{tot} = \sum C_{i,esm} \quad (3.6)$$

$$C_{tot} = \sum C_{i,vol} \quad (3.7)$$

where C_{tot} = soil organic carbon density using either the volume based method or equivalent soil mass method.

The forest floor carbon density was calculated using equation (3.8).

$$C_{ff} = FF \times FFC \quad (3.8)$$

where C_{ff} = forest floor carbon density (FFCD, Mg C ha^{-1}), *FF* = forest floor mass (Mg ha^{-1}), *FFC* = forest floor carbon concentration (%).

A collection of the terms, symbols, abbreviation and units that are referred to throughout the study is presented in table 3.1.

Table 3.1. Collection of the terms, symbols, abbreviations and units used throughout this study.

Quantity	Symbol	Abbreviation	Units
Soil Organic Carbon	$conc_i$	SOC	%
Bulk Density	ρ_d	BD	g cm^{-3}
Dry soil mass	M_i	M	Mg ha^{-1}
Depth of Soil	h	DS	cm
Soil Organic Carbon Density (volume based method)	$C_{i,vol}$	SOCDvol	Mg C ha^{-1}
Soil Organic Carbon Density (equivalent soil mass method)	$C_{i,esm}$	SOCDesm	Mg C ha^{-1}
Soil Organic Carbon Density (either Equivalent Soil Mass Method or volume based method)	C_{tot}	SOCD	Mg C ha^{-1}
Forest Floor Mass	FF	FFM	Mg ha^{-1}
Forest Floor Carbon Concentration	FFC	FFCC	%
Forest Floor Carbon Density	C_{ff}	FFCD	Mg C ha^{-1}
Stocking Density	ST	STDE	stem ha^{-1}
Diameter at Breast Height	d	DBH	Cm
Stem and Branch Biomass	S	SB	kg
Leaf Biomass	L	LB	kg
Root Biomass	R	RB	kg
Aboveground Biomass Carbon Density	C_a	AGCD	Mg C ha^{-1}
Belowground Biomass Carbon Density	C_w	BGCD	Mg C ha^{-1}
Biomass Carbon Density	C_b	BCD	Mg C ha^{-1}
Ecosystem Carbon Density	C_e	ECD	Mg C ha^{-1}

i = i^{th} depth (i =1,2,3 corresponding to the 0-10, 10-20, 20-30 and 0-30 cm soil depths)

3.2. Peatland methodology

3.2.1. Site selection

The peatland sites were randomly chosen from the filtered NFI list used to select the paired plot sites (Chapter 3.1.1.). Peat soils were defined as having a peat depth greater than 30 cm and having a soil organic matter content greater than 30% (Hammond, 1981). Twenty-four sites were selected randomly from the groups which included conifer raised bog, conifer high level blanket bog and low level blanket bog (Table 5.1). A number of sites were inappropriate for sampling due to being harvested or not meeting the definition of peat, and so these sites were replaced with sites from the back up list. These sites were again split up and sampled by ForestC (14 sites) and CARBiFOR II (10 sites). The peat sites were not sampled with the paired plot methodology due to the large variation of soil organic carbon density within each site, which would mask any changes in the soil organic carbon density following afforestation.

3.2.2. Field sampling

The sampling design was adapted from Davis et al. (2004). Each NFI site was located using GPS (GPS 60 Garmin, USA). At each site a 20 m by 20 m square plot was set out, and then partitioned into four 10 m by 10 m quadrants. Within each quadrant, 4 points were randomly selected and the soil sampled using a Russian peat corer (Eijkenkamp Agrisearch Equipment BV, Netherlands) with a volume of 500 cm^3 for bulk density (g cm^{-3}) and SOC (%). The peat was sampled over its full depth from the peat surface vertically down the profile in increments of 50 cm. A second sample (50 to 100 cm) was taken at a point 10 cm west of the first sampling point to avoid the effects of compaction from the previously extracted sample and a third sample (100 to 150 cm) was taken 10 cm away from the second sampling point, while the fourth (150 to 200 cm) reverted back to the original sampling point and so on. When the bottom of the peat was reached, indicated by the presence of a mineral layer or the presence of impenetrable rock, the final depth was recorded and the sample taken. If a sample was seen to contain a peat pipe or cavity, its depth was recorded with the sample ID. Each sample was placed into a labeled, sealable polythene bag in the field. At sites RB1-5, HLB1-4 and LLB1-5 around each sampling point, 4 additional points were used to determine the peat depth using the Russian peat corer. Each point was located 50 cm north, south, east or west of the initial sampling point. This was repeated for each of the 4 random points within the 4 quadrants of the site.

There were limitations to sampling using the Russian peat corer in the top layer due to the inability of the corer to cut through the roots of the ground cover vegetation, usually most prominent in the top 20 cm. To compensate for this, a random point was chosen within each 10 m by 10 m quadrant where a hole was dug and samples were taken using stainless steel bulk density rings (Eijkenkamp Agrisearch Equipment BV, Netherlands) of 8 cm diameter by 5 cm height. Samples were taken with the rings to replace those that the Russian peat corer could not take reliably. If the Russian corer could not reliably sample 0–30 cm, we used the rings to sample at depths: 0-5 cm; 5-10 cm; 15-20 cm; and 25-30 cm (5 cm gaps were placed between samples after 10 cm to avoid the effects of compaction from previous samples).

To summarise, at each of the 24 sites we obtained: (1) 16 peat profiles using the Russian peat corer for bulk density and SOC laboratory measurements; (2) four surface profiles (typically 0-30 cm) using stainless steel rings for bulk density and SOC: and (3) 80 individual estimates of peat depth, using the Russian auger at sites RB1-5, HLB1-4, LLB1-5, and 16 individual estimates of peat depth at sites RB6-11, HLB5-6 and LLB6-7. The distribution of samples within the 20 m x 20 m plot can be seen in Figure 3.2.

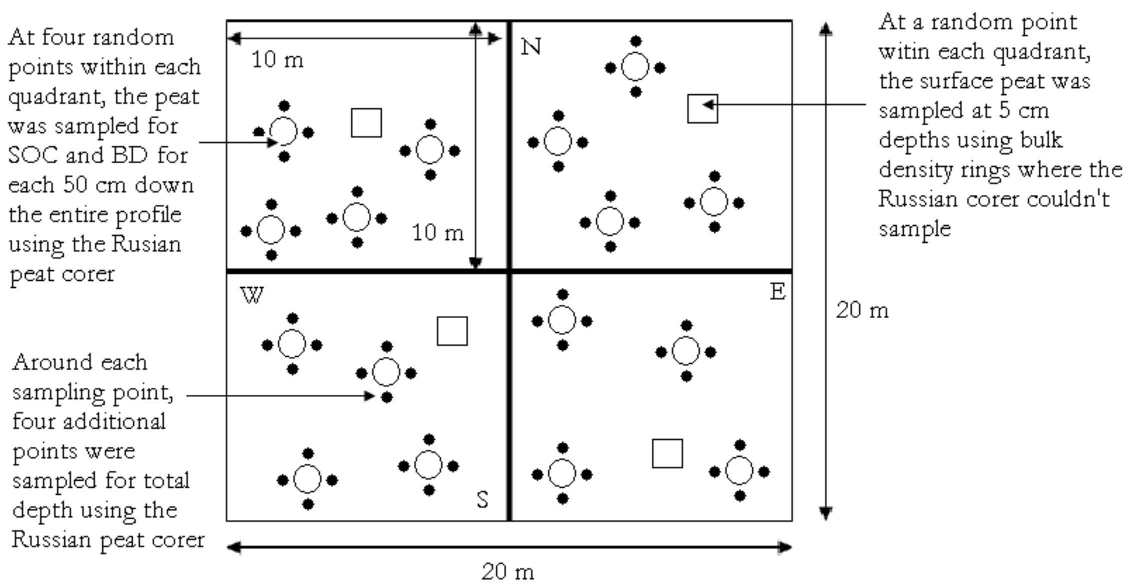

Figure 3.2. The sampling distribution within the 20 m x 20 m plot used at each peatland site.

3.2.3. Sample analysis

The peat samples were analysed in the same way for soil organic carbon content as the mineral soil in chapter 3.1.3. Equations 3.1 and 3.3 were used to calculate the bulk density and soil organic carbon density of each depth. While equation 3.6 was used to calculate the total soil organic carbon density of the peat soil over the total depth.

3.3. Ash chronosequence methodology

3.3.1. Site selection

Sites for the chronosequence study were found upon consultation with colleagues and relevant forest managers. The sites, G0 (grassland site representative of the pre-afforestation land use) and F12 (12 year old ash stand) were located on private land within Co. Offaly, Ireland, while sites F20, F27, F40 and F47 (20, 27, 40 and 47 year old ash stands respectively) are located within the Coillte Teoranta owned state forest of Castlemorris in Co. Kilkenny, Ireland. The chronosequence technique used in this study allows for the integration of independent forest stands with different ages, into one unit, substituting space for time. The stands are as close to identical in all aspects other than stand age (Taylor et al., 2007; Hedde et al., 2008; Black et al., 2009; Tang et al., 2009).

3.3.2. Field sampling

The soil and forest floor of each chronosequence site was sampled using the same methodology as that of the paired plot study (Chapter 3.1.2.). The methodology used to measure the above ground biomass of the sites was adapted from the NFI Protocol (NFI, 2007b) and was non-destructive. Three random points were chosen within each forest site and a circular plot of 500 m^2 (25.24 m diameter) was set up with random points as plot centers. The diameter at breast height (DBH, cm, at 1.3 m) was recorded for each tree that had a DBH greater than 7 cm within each plot. Sites F40 and F47 were smaller than the other ash stands (<1500 m^2) and so the third plot was set to an area of 250 m^2.

3.3.3. Sample analysis

The soil of each chronosequence site was analysed for soil organic carbon concentration and soil texture using the same methodology as that of the paired plot study (Chapter 3.1.3.). Sites G0 and F40 were found to have carbonates present. Equation 3.1 was used to calculate the soil bulk density. Equation 3.3 was used to measure the SOCD of each soil depth of the grassland site using the volume-based method and equation 3.5 was used to calculate the SOCD of each soil depth for each forest site using the equivalent soil mass method. The SOCD (0-30 cm) of the forest sites were calculated using equation (3.6) and the SOCD of the grassland site was calculated using equation (3.7). The forest floor samples were dried at 55 °C to achieve a constant dry weight. The stem and branch biomass was calculated from an allometric equation (3.9) of ash trees within England (Bunce, 1968).

$$ln(S) = -2.4598 + 2.4882ln(d) \tag{3.9}$$

where S = stem and branch biomass (kg); d = DBH (cm)

The leaf biomass data were calculated from equation (3.10) of ash trees from an Italian study (Alberti et al., 2005).

$$L = 0.003d^{2.31} \tag{3.10}$$

where L = leaf biomass (kg); d = DBH (cm)

The root biomass was calculated from equation (3.11) (Cairns et al., 1997).

$$R = 0.26b \tag{3.11}$$

where R = root biomass (kg), b = aboveground biomass (kg).

The biomass and forest floor mass values were multiplied by 0.5 to estimate the mass of C (IPCC, 1997). The forest floor carbon density was calculated using equation (3.12).

$$C_{ff} = 0.5FF \tag{3.12}$$

where C_{ff} = forest floor carbon density (FFCD, Mg C ha^{-1}), *FF* = forest floor mass (Mg ha^{-1}).

The carbon density of the above-ground biomass was calculated using equation (3.13).

$$C_a = (S \times ST + L \times ST) \times 0.5 \times 10^{-3} \tag{3.13}$$

where C_a = aboveground biomass carbon density (AGCD, Mg C ha^{-1}), *ST* = stocking density (stem ha^{-1}), 10^{-3} is a conversion factor.

The carbon density of the below-ground biomass was calculated using equation (3.14).

$$C_w = 0.26 C_a \tag{3.14}$$

where C_w = belowground biomass carbon density (BGCD, Mg C ha^{-1}).

The carbon density of the biomass was calculated using equation (3.15).

$$C_b = C_a + C_w \tag{3.15}$$

where C_b = biomass carbon density (BCD, Mg C ha^{-1}),

The carbon density of the ecosystem was calculated using equation (3.16).

$$Ce = C_b + C_{ff} + C_{tot} \tag{3.16}$$

where Ce = ecosystem carbon density (Mg C ha^{-1}).

4. What is the impact of afforestation on the carbon stocks of Irish mineral soils?

Michael L. Wellock[1*], Christina M. LaPerle[1] & Gerard Kiely[1]

[1] *Centre for Hydrology, Micrometeorology and Climate Change, Department of Civil and Environmental Engineering, University College Cork, Cork, Ireland.*

Submitted to: Forest Ecology and Management on the 3/4/2011. Has been accepted for publication pending corrections.

Keywords: Afforestation; Soil; Carbon; Sequestration; Paired Sites.

4.1. Abstract

Ireland has implemented a large afforestation program in recent decades, with much of this taking place since the mid 1980s. This presents Ireland with the opportunity to offset carbon emissions through carbon sequestration in forests, as the latter are known to sequester a large amount of carbon into the tree biomass. However, the effects of afforestation on soil carbon in the Irish humid temperate climate are not well understood. In this study we use the paired-plot methodology (Davis and Condron, 2002) to assess the impact of afforestation on the soil organic carbon density (SOCD) of 21 pairs of sites across Ireland. We focussed on the SOCD change between adjacent forested and non forested sites, considering the seven variables: forest type; pre-afforestation land use; soil texture; soil pH; soil cultivation disturbance; precipitation; and forest age. Excluding the forest floor carbon density, we found that afforestation of Irish soils to a depth of 0-30 cm resulted in a small, but not significant, loss of SOCD from the soil. However, when the forest floor carbon density was added to the SOCD, we found that conifer afforestation led to a small, but not significant, increase in ecosystem carbon density. We found that to sequester C into the soil, broadleaf species are better than coniferous. Furthermore, the loss of SOCD was minimised in afforestation on rough grazing sites or on soils with low pH. However, the low number of sites within the study is a source of uncertainty and more work must be done to assess SOCD change before any firm conclusions can be made. This work provides baseline data and future work estimating SOCD changes due to land use or management changes should use the equivalent soil mass (ESM) correction method instead of the volume based method. The latter can over- or underestimate SOCD change due to variability in soil bulk density between sites. The large afforestation programmes to be implemented in Ireland in the next decade provides an opportunity to greatly improve estimates of Irish SOCD change. We suggest implementing a large number of resampling studies, measuring the change in SOCD following afforestation for a number of factors for a number of years.

4.2. Introduction

At the beginning of the twentieth century forests covered only 1% of the total Irish land area (Pilcher and Mac an tSaoir, 1995). However, due to the efforts of successive governments there has been rapid afforestation since the 1960s resulting in a 10.0% forest land cover as of 2007

(NFI, 2007a). A large proportion of this afforestation took place after the mid-1980s, encouraged by government grant incentives targeted at private landowners. Consequently, 63% of Irish forests are less than 20 years old (NFI, 2007a), providing Ireland the opportunity to contribute to meeting its international obligations set forth by the United Nations Framework Convention on Climate Change (UNFCCC, 1992). These obligations include the limitation of greenhouse gas emissions to 13% above 1990 levels. In order to promote accountability for these commitments, the UNFCCC treaty and the Kyoto Protocol (Kyoto Protocol, 1997) mandate signatories to publish greenhouse gas (GHG) emissions inventories for both greenhouse gas sources and sinks. Article 3.3 of the Kyoto Protocol allows changes in carbon (C) stocks due to afforestation, reforestation, and deforestation since 1990 to be used to offset inventory emissions. Therefore, due to the rapid rate of afforestation and the increased ecosystem (biomass, forest floor and soil) carbon sequestration since 1990, Ireland's recent afforestation programme has the potential to significantly offset some of its GHG emissions.

It is well established that there is a large increase in the carbon stored in the biomass following afforestation (Morris et al., 2007; Black et al., 2009; Mendoza-Ponce and Galicia, 2010), but the effects on soil organic carbon (SOC) are still uncertain (Kirschbaum et al., 2008). Soils contain approximately two-thirds of the C stored within forest ecosystems (Dixon et al., 1994). The residence time of stable fractions of SOC can be > 1000 years (von Lutzow et al., 2006) making it a much more stable sink than living plant biomass (Laganière et al., 2010). In addition to being able to estimate the carbon in tree biomass, it is therefore vital to measure the change in SOC stocks following afforestation, and to determine the mechanisms involved in controlling SOC dynamics.

Soil C stocks are determined by the balance between the inputs of C through litterfall and rhizodeposition and the loss of C mainly through soil organic matter (SOM) decomposition (Jandl et al., 2007). The change in soil C following afforestation is controlled by a number of factors, including: previous land use (grasslands, cropland, etc); tree species; soil cultivation method; soil properties (clay content, pH); stand age; site management; topography and climate (Guo and Gifford, 2002; Paul et al., 2002; Jandl et al., 2007; Laganière et al., 2010). A number of studies have been conducted to investigate afforestation induced SOC changes with different conclusions. Some studies have found no change in SOC stocks after afforestation (Bashkin and Binkley, 1998; Chen et al., 2000; Davis, 2001; Davis et al., 2003; DeGryze et al., 2004; Davis et al., 2007; Smal and Olszewska, 2008; Peri et al., 2010). Others have found an increase in SOC stocks

(Post and Kwon, 2000; Guo and Gifford, 2002; Del Galdo et al., 2003; Hooker and Compton, 2003; Grünzweig et al., 2007; Morris et al., 2007; Black et al., 2009; Mao et al., 2010) while some have found a decrease in SOC stocks (Parfitt et al., 1997; Perrott et al., 1999; Ross et al., 1999; Chen et al., 2004; Farley et al., 2004; Alberti et al., 2008; Wellock et al., 2011a). However, several studies have shown a similar trend where initially there is a reduction in SOC stocks as the decomposition of soil organic matter is greater than the input of organic matter from the trees. Over time the soil organic matter input increases with the productivity of the forest stands and the soils switch from being a C source to a C sink. This can lead to an eventual recovery of soil C to the pre-afforestation levels and in some situations surpass them (Romanyà et al., 2000; Davis and Condron, 2002; Paul et al., 2002; Turner et al., 2005; Zerva et al., 2005; Wang et al., 2006; Huang et al., 2007; Ritter, 2007; Hu et al., 2008; Laganière et al., 2010; Mao et al., 2010).

A number of reviews have been conducted to examine the effects of afforestation on soil C stocks globally (Guo and Gifford, 2002; Paul et al., 2002; Laganière et al., 2010) and for specific countries (Davis and Condron, 2002). The latter, in a review of paired plot (adjacent plots on same soils, one afforested, and one as original land use) studies of coniferous forestry in New Zealand found an initial loss of soil C, before recovery to pre-afforestation levels after 20 years. Guo and Gifford (2002) in a meta-analysis of different land use changes on soil C reported a 10% decline in SOC with afforestation of pasture land and a 13% reduction from conversion of native forest to plantation forestry. It was found that when pasture and native forests were afforested with broadleaf, there was little change; however, there was a significant loss in soil C following afforestation with pine (Guo and Gifford, 2002). Paul et al. (2002), in a review of 43 afforestation studies found the key factors in order of importance to be: previous land use; climate; and the type of forest. Laganière et al. (2010) published a meta-analysis (synthesised from 33 publications) of the impacts of afforestation on the soil C stocks of agricultural land. They found that the main factors effecting soil C change to be, in order of importance: previous land use; tree species; soil clay content; pre-planting disturbance; and to a lesser extent, climatic zone.

Two studies have examined the impact of afforestation on the C stocks of mineral soils in Ireland, with different conclusions. Black et al. (2009) analysed a chronosequence of Sitka spruce stands on surface-water gley soils in Co. Wicklow, and found an increase in soil C stocks from the pre-afforestation grassland site to the 9 year old spruce site, and further increases with stand age. Wellock et al. (2011a) in an ash forest chronosequence located on brown earth soils in the Irish midlands, observed a continuous decline in SOCD from the pre-afforestation grassland sites with stand age up to the 27 year old stand. Thereafter, the SOCD began to increase. Although the soil begins to sequester C (after age 27), it did not accrue C as quickly, as it was initially released from the soil, with the result that the SOCD of the 47 year old ash forest were only 79% of pre-afforestation grassland levels. The differing conclusions of these two Irish studies show that there is still not a consensus on the impacts of afforestation on Irish soil C stocks and the controlling factors, suggesting that further research is required.

The most commonly used method to determine SOCD is the volume-based method which multiplies the SOC concentration (%) by bulk density (g cm^{-3}) to a fixed depth. The volume-based method in assessing the effects of land use change on SOCD assumes that soil bulk density is static through time (Markewitz et al., 2002; Sartori et al., 2007; Mao et al., 2010). However, soil bulk density does vary spatially and temporally (Amador et al., 2000; Gifford and Roderick, 2003; Kulmatiski and Beard, 2004; Lee et al., 2009; Wuest, 2009). The volume-based method does not account for variations in soil mass and this introduces uncertainty in SOCD changes (Ellert and Bettany, 1995; Markewitz et al., 2002; Murty et al., 2002; Lee et al., 2009; Mao et al., 2010). The equivalent soil mass (ESM) correction was proposed by Ellert and Bettany (1995) as a more reliable method to determine changes in SOCD due to changes in land use or management practices and a number of studies have used it since (Ellert and Bettany, 1995; Yang and Wander, 1999; Gifford and Roderick, 2003; VandenBygaart, 2006; VandenBygaart and Angers, 2006; Lee et al., 2009; Wuest, 2009; Mao et al., 2010). The ESM is defined as the reference soil mass per unit area chosen in a layer and the equivalent C mass (ECM) is the C stored in an ESM (Ellert et al., 2001; Mao et al., 2010).

The objectives of this paired plot study were: (1) to quantify the forest floor carbon density and soil organic carbon density (0-30 cm depth) of 21 forest sites and their 21 adjacent non-forest site on same soils; and (2) to assess the impacts of afforestation on SOCD using the paired site method.

4.3. Materials and methods

4.3.1. Site Selection

The forest sites were selected from stands surveyed for the Irish National Forest Inventory (NFI) which systematically sampled 1,742 forest plots across the Republic of Ireland (NFI, 2007b). We filtered the NFI database and rejected those sites that were inaccessible, reforested, and younger than 15 years or had soil types that were representative of less than 1% of Irish forest soils (regosols, lithosols and rendzinas). All sites with peat soils were removed from this study and were sampled separately from the mineral soils and are presented in Chapter 5, (Wellock et al. 2011b). The remaining 98 sites were sorted into 3 forest categories: conifer, broadleaf and mixed. These sites were then further sub-divided by soil type; brown earth, gley, podzol and brown podzolic. Those categories with 6 or more sites were then selected. From those categories sites were randomly chosen from each group (of soil type and forest type) for sampling. Any sites left over, were retained as replacement sites. To assess the impact of afforestation, all the selected 21 forest sites were paired with an adjacent non-forest site that had the same current land use as the forest site had prior to afforestation. The non-forest site was selected so that it had the same attributes (relief, aspect, elevation, soil type etc) as the forest site, with the only difference being the current land use. The details and locations of sites are shown in Table 4.1 and Figure 4.1.

Table 4.1. Characteristics of each site; F, forest site; NF, non-forest site; DMB, double mould board ploughing; SMB, single mould board ploughing; plough, agricultural ploughing. Pit planting, planting of trees into a pit opened with a spade, no mechanical site preparation is used.

Site	Forest Type	Soil Type	Previous Land Use	Elevation (F/NF) (m)	Site Preparation	Precipitation (mm yr^{-1})	Forest Age (years)	Georeference Position
BLBE1	Broadleaf	Brown Earth	Pasture	97/97	Pit Planting	911.9	42	52° 54' N, 7° 22' S
BLBE2	Broadleaf	Brown Earth	Pasture	108/108	Plough	992.7	20	52° 15' N, 8° 33' S
BLBE3	Broadleaf	Brown Earth	Pasture	44/38	DMB	1198	17	52° 46' N, 8° 25' S
CBE1	Conifer	Brown Earth	Pasture	170/169	Mounding	1083.5	19	53° 9' N, 8° 32' S
CBE2	Conifer	Brown Earth	Pasture	188/185	Pit Planting	1154.7	34	52° 57' N, 6° 37' S
CBE3	Conifer	Brown Earth	Scrub	120/100	Pit Planting	1246.6	42	52° 24' N, 7° 33' S
CG1	Conifer	Gley	Rough Grazing	199/207	Mounding	883.5	17	52° 52' N, 7° 11' S
CG2	Conifer	Gley	Scrub	317/311	Plough	1073.7	33	52° 50' N, 8° 30' S
CG3	Conifer	Gley	Rough Grazing	178/180	SMB	1409.9	29	52° 10' N, 7° 47' S
CG4	Conifer	Gley	Rough Grazing	109/108	DMB	1213.3	22	53° 47' N, 8° 23' S
CP1	Conifer	Podzol	Rough Grazing	212/192	Mounding	1183.5	18	52° 44' N, 8° 19' S
CP2	Conifer	Podzol	Rough Grazing	326/325	DMB	1555.1	41	52° 1' N, 8° 59' S
CP3	Conifer	Podzol	Rough Grazing	235/240	Pit Planting	1154.7	68	52° 15' N, 7° 56' S
CP4	Conifer	Podzol	Rough Grazing	255/279	Plough	1008.2	22	52° 16' N, 8° 31' S
CBP1	Conifer	Brown Podzolic	Rough Grazing	293/307	Pit Planting	1307.7	68	53° 3' N, 7° 38' S
CBP2	Conifer	Brown Podzolic	Pasture	314/304	Pit Planting	996.8	22	52° 43' N, 7° 8' S
MBE1	Mixed	Brown Earth	Pasture	55/55	Pit Planting	964.4	67	53° 35' N, 8° 12' S
MBE2	Mixed	Brown Earth	Pasture	117/104	Pit Planting	785.7	51	53° 0' N, 7° 6' S
MBE3	Mixed	Brown Earth	Pasture	100/108	SMB	1038.3	33	52° 30' N, 7° 1' S
MG1	Mixed	Gley	Scrub	53/46	Plough	1084.8	37	52° 48' N, 8° 46' S
MG2	Mixed	Gley	Rough Grazing	60/57	Mounding	2301.9	17	51° 54' N, 9° 23' S

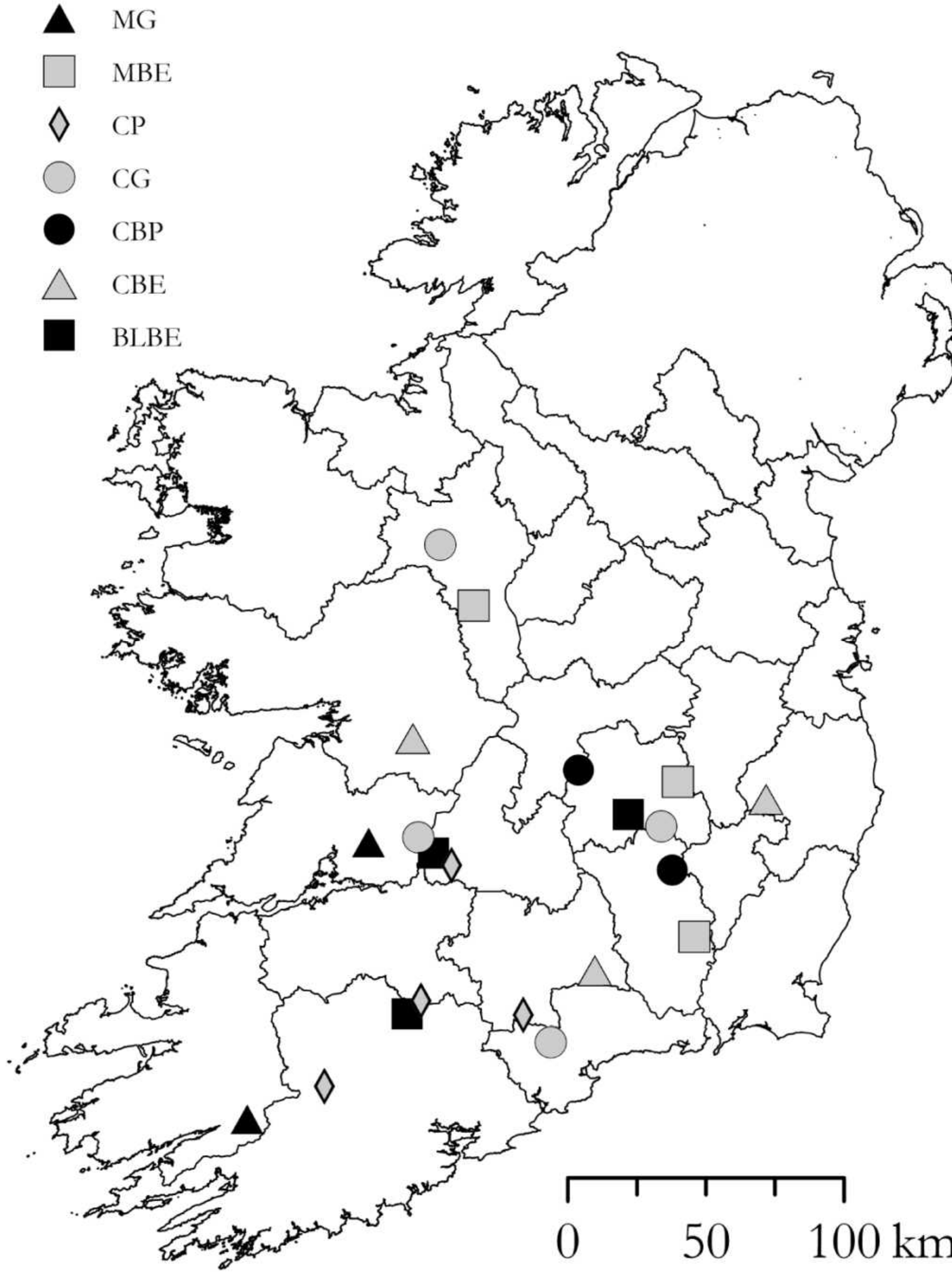

Figure 4.1. Locations of all 21 sites within Ireland arranged by forest type and soil type. MG, mixed forest gley soil; MBE, mixed forest brown earth soil; CP, conifer forest podzol soil; CG, conifer forest gley soil; CBP, conifer forest brown podzol soil; CBE, conifer forest brown earth soil; BLBE, broadleaf forest brown earth soil.

4.3.2. Sampling

Field sampling took place between October 2007 and May 2009 and the sampling design was adapted from Davis et al. (2004). At both the forest and adjacent non-forest sites, a 20 m x 20 m square plot was established and divided into four, 10 m x 10 m quadrants. The soil within each site was sampled to a depth of 30 cm. Within each quadrant at a preselected random point, a soil pit was dug and sampled for soil classification and bulk density. Bulk density samples were taken at depths of 0-5, 5-10, 15-20 and 25-30 cm using stainless steel bulk density rings (Eijkenkamp Agrisearch Equipment BV, Netherlands) of 8 cm diameter by 5 cm height. Around the pit, 8 points were sampled using a soil auger (Eijkenkamp Agrisearch Equipment BV, Netherlands) to depths of 0-10, 10-20 and 20-30 cm and analyzed for soil organic carbon (SOC, %).

The forest floor was sampled at all forest sites along with the scrub non-forest sites, however, the organic layer of the pasture and rough grazing non-forest sites were not sampled. At three points within each quadrant, the forest floor was sampled from 0.1 m^2 square plots. Each forest floor sample was separated into 3 separate classes: (1) fine woody debris (woody material with a diameter >2.5cm and <7.5 cm); (2) litter (non decomposed material of diameter <2.5 cm); and (3) F/H layer (decomposed material that is not mixed with the soil).

All soil and forest floor samples were stored at 4 ^oC before being oven-dried at 55 ^oC until a constant dry weight was achieved. The soil samples taken with the soil auger for SOC were sieved to <2 mm (the 8 soil samples were bulked for each depth for each quadrant by equal volume). All soil samples were tested for carbonates. Any samples that tested positive for carbonates were treated with sulphurous acid to remove the carbonates (Nelson and Sommers 1996). The soils of sites CBE1, CG4, MBE1 and MBE2 were treated for carbonates. The forest floor and sieved soil samples were ground to a fine powder before combustion in a C/N analyser (Elementar - Vario Max CN) to determine SOC concentration (%). The soil texture was analysed at each depth for all sites from the bulked soil samples, using the hydrometer method, ASTM D422 (2002), and measured for pH using 1:1 soil to water using a LabFit AS-3000 pH analyser (McLean, 1982). The bulk density soil samples were sieved to 2 mm, with both the <2 mm fraction and >2 mm fractions being stored separately. The bulk density was estimated using equation (4.1),

$$\rho_d = \frac{MA}{SV - CFV} \tag{4.1}$$

where ρ_d = bulk density (g cm^{-3}); MA = mass of dry sample <2 mm (g); SV = sampler volume (cm^3); CFV = >2 mm coarse fraction volume (cm^3)

The bulk density samples were converted to 10 cm depths to match the SOC data. The soil mass was calculated using equation (4.2).

$$M_i = \rho_{di} \times h \times 10^4 \tag{4.2}$$

where M_i = dry soil mass (Mg ha^{-1}) at the i^{th} depth (i =1,2,3 corresponding to the 0-10, 10-20 and 20-30cm depths); h = depth of soil layer (m); 10^4 = unit conversion factor (m^2 ha^{-1}).
The soil organic carbon density of the soil to a fixed depth was calculated using equation (4.3).

$$C_{i,vol} = conc_i \times M_i \tag{4.3}$$

where $C_{i,\mathrm{vol}}$ = soil organic carbon density, volume based method (SOCDvol, Mg C ha^{-1}); conc$_i$ = SOC (%)

The equivalent soil mass method uses the soil mass of each soil layer sampled at the non-forest site as the ESM for the layer, using equation (4.4), the equivalent carbon mass (ECM) can be calculated using equation (4.5) (Lee et al., 2009).

$$M_{i,add} = M_{i,equiv} - M_i \tag{4.4}$$

$$C_{i,esm} = C_{i,vol} - conc_{i+1} \times M_{i-1,add} + conc_{i-1} \times (M_{i,add} - M_{i-1,add}) \tag{4.5}$$

where $C_{i,\mathrm{esm}}$ = the soil organic carbon density, equivalent soil mass method (SOCDesm, Mg C ha^{-1}); $M_{i,\mathrm{equiv}}$ = non-forest soil mass (Mg ha^{-1}); $M_{i-1,\mathrm{add}}$ and $M_{i,\mathrm{add}}$ = the additional soil masses that are used to estimate the ESM (Mg ha^{-1}); $conc_{i-1}$ and $conc_{i-1}$ = SOC concentration (%) for the additional soil mass determined by the location of soil mass used for the corrections between the soil layers at each sampling time. For more information on the ESM method see Ellert and Bettany (1995), Ellert et al. (2001) and Lee et al. (2009). The soil organic carbon density (0-30 cm) of the forest sites were calculated using equation 4.6 and the soil organic carbon density of the non-forest site was calculated using equation 4.7.

$$C_{tot} = \sum C_{i,\,esm} \tag{4.6}$$

$$C_{tot} = \sum C_{i,\,vol} \tag{4.7}$$

where C_{tot} = soil organic carbon density (0-30 cm) using either the volume based method or equivalent soil mass method.

The forest floor carbon density was calculated using equation (4.8).

$$C_{ff} = FF \times FFC \tag{4.8}$$

where C_{ff} = forest floor carbon density (FFCD, Mg C ha^{-1}), *FF* = forest floor mass (Mg ha^{-1}), *FFC* = forest floor carbon concentration (%).

To summarise, at each of the 21 sites we obtained for both the forest and paired site: (1) 32 samples for SOC at each depth of 0-10, 10-20 and 20-30 cm using the soil auger; (2) 4 samples of bulk density at each depth of 0-5, 5-10, 15-20 and 25-30 cm using the stainless steel rings; and (3) 12 samples of fine woody debris, litter and F/H layer of the forest floor.

4.3.3. Data presentation and statistical analysis

The 21 sites were analysed for 7 variables that are considered to influence soil organic carbon density. These include: forest type (conifer, mixed, broadleaf); pre-afforestation land-use (pasture, rough grazing, scrub); soil texture (loam, sandy loam, clay loam, silt loam); soil pH (<5, 5-7); cultivation disturbance (low-level, high/medium-level); mean annual precipitation (750-1000, 1001-1250, 1251-1500, 1501-2000, 2001-2500 mm) and the forest age (10-19, 20-29, 30-39, >40 years). The SOC, bulk density and SOCD values of the forest and non-forest sites were compared along these groupings.

The data for the bulk density, SOCD and the forest floor carbon density were normally distributed, and so the mean carbon densities of each forest and non-forest site were compared for significant differences using the paired *t*-test. The data for the soil organic carbon were not normally distributed and so were analysed by the Wilcoxon signed-rank test. Differences between the mean forest floor carbon densities were analysed by ANOVA, and used the games-howell post hoc test as the variance was not homogenous. All statistical analysis was carried out using SPSS (SPSS Inc., SPSS Statistics, Student Version, Release 17.0, 2008).

4.4. Results

4.4.1. Change in soil properties following afforestation

Among all 21 sites, the SOC (0-30 cm) of the non-forest sites had a mean value of 5.0 ± standard deviation of 2.1% compared to 4.7 ± 2.6% at the forest sites. The bulk density (0-30 cm) of the 21 sites increased following afforestation, with a non-forest mean value of 0.88 ± 0.20 g cm^{-3} compared to 0.94 ± 0.20 g cm^{-3} for the forest sites. Amongst all sites the mean soil organic carbon density (SOCD, 0-30 cm) for the non-forest sites was higher at 111.7 ± 30.8 Mg C ha^{-1} compared to 106.0 ± 29.2 Mg C ha^{-1} for the forest sites. When the forest floor carbon densities were added to the SOCD values the total C in the forests increased to 112.8 ± 28.2 Mg C ha^{-1} (slightly more than the 112.1 ± 30.3 Mg C ha^{-1}) in the non-forests.

4.4.2. Factors affecting soil organic carbon density change following afforestation

4.4.2.1. Forest type

The 21 sites were separated by forest type which comprised: 13 conifer sites; 5 mixed sites; and 3 broadleaf sites. Conifer and broadleaf were defined when the dominant forest type represented greater than 81% of the canopy cover, while the mixed forests were a combination of both conifer and broadleaf species where the lesser forest type makes up at least 20% of the canopy cover (NFI, 2007b).

There were no significant differences between the bulk densities (0-30 cm) of the forest and non-forest sites for the conifer and mixed forest types. The bulk density of the conifer and mixed forests tended to increase slightly following afforestation, while the bulk density of the broadleaf sites declined. The conifers saw a 10.7% increase from the non-forest value of 0.82 ± 0.17 g cm^{-3} to the forest value of 0.90 ± 0.21 g cm^{-3}. The bulk densities of the mixed forests tended to increase slightly after afforestation by 9.5% from the non-forest value of 0.91 ± 0.23 g cm^{-3} to the forest value of 1.00 ± 0.23 g cm^{-3}, while the bulk densities of the broadleaf forests significantly decreased after afforestation by 14.4% from the non-forest value of 1.13 ± 0.07 g cm^{-3} to the forest value of 0.97 ± 0.02 g cm^{-3} (see Figure 4.2a).

There were no significant differences between the SOC (0-30 cm) of the forest and non-forest sites for each forest type. The SOCs of the three forest types show the opposite trend to the bulk density values. The SOC of the conifer sites decreased by 5.3% from the non-forest value of 5.4 ± 1.9%, to the forest value of 5.1 ± 3.0%. The SOC of the mixed sites decreased by 14.1% from the pre-afforestation level of 5.1 ± 3.2% to the forest value of 4.4 ± 2.4%. The broadleaf SOC increased by 9.1% from the non-forest value of 3.4 ± 0.7% to the forest site value of 3.7 ± 0.8% (see Figure 4.2b).

There were no significant differences between the SOCD (0-30 cm) of the forest and non-forest sites for each forest type. The SOCD values matched the trends in SOC between forest types. The broadleaf forests showed an increase of 3.9% from the pre-afforestation mean value of 111.2 ± 17.2 Mg C ha^{-1} to the mean forest SOCD of 115.5 ± 19.3 Mg C ha^{-1}. The SOCD of the conifer forests decreased by 6.6% from the pre-afforestation mean value of 110.0 ± 33.3 Mg C ha^{-1} to the mean forest SOCD of 102.8 ± 33.1 Mg C ha^{-1} and the SOCD of the mixed forests also decreased by 6.4% from the non-forest mean value of 116.2 ± 34.9 Mg C ha^{-1} to the mean forest SOCD of 108.7 ± 26.0 Mg C ha^{-1} (see Figure 4.2c).

The forest floor carbon density varied between the 3 forest types, with the 9.3 ± 5.1 Mg C ha^{-1} of the conifer sites significantly larger ($p<0.05$) than the 3.2 ± 2.3 Mg C ha^{-1} and 1.7 ± 1.0 Mg C ha^{-1} forest floor carbon densities of the mixed and broadleaf forest floors, respectively. When the forest floor carbon density is added to the SOCD the coniferous forests switched from losing to sequestering C, a 1.5% increase to 112.1 ± 32.7 Mg C ha^{-1} at the forest site compared to 110.4 ± 32.6 Mg C ha^{-1} at the non-forest sites (see Figure 4.2d). The mixed forest sites still lost C with the forest floor carbon density addition, although the loss of C was smaller, a 1.5% from a non-forest value of 117.0 ± 35.0 Mg C ha^{-1} to the forest value 111.9 ± 24.2 Mg C ha^{-1}. As the broadleaf forests had the smallest forest floor carbon density, the percentage change in C only increased by a small amount to 5.4% larger than the pre-afforestation value of 111.2 ± 17.2 Mg C ha^{-1} to the forest mean value of 117.1 ± 18.4 Mg C ha^{-1}. However, the forest sites were not significantly different than their adjacent non-forest site when the forest floor carbon density was added to the SOCD.

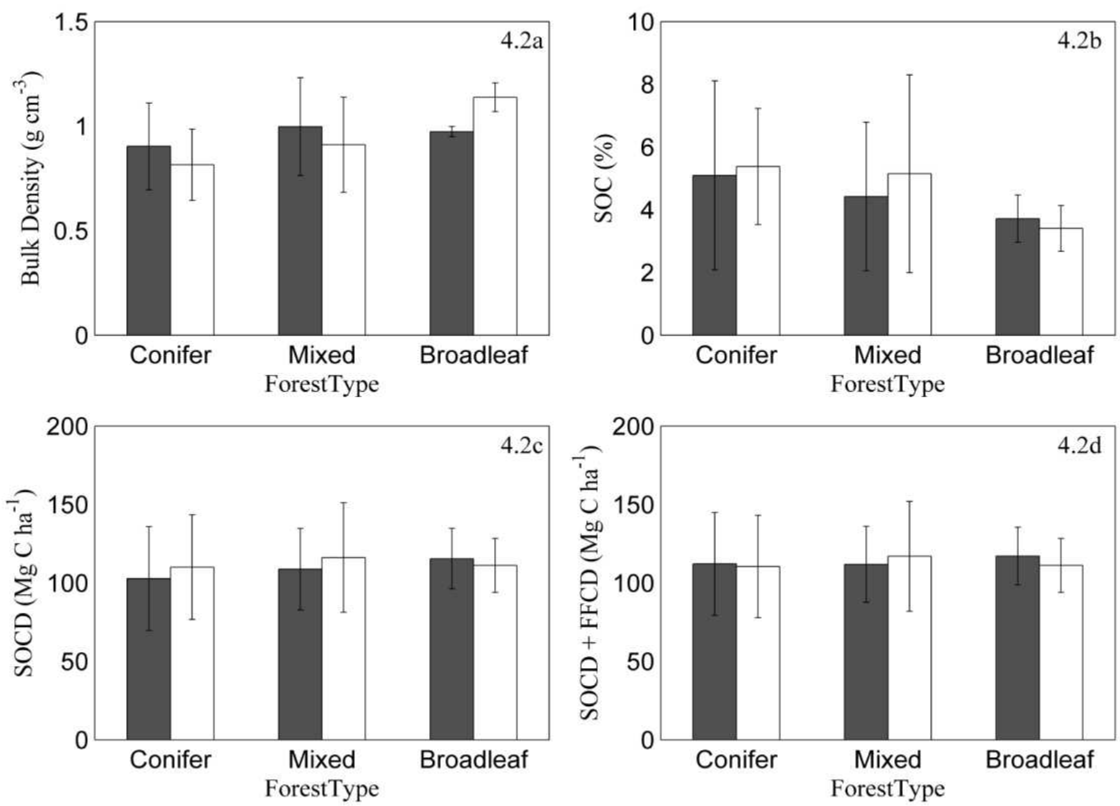

Figure 4.2a. The mean forest bulk density (0-30 cm, filled bar, g cm^{-3}) and non-forest site bulk density (0-30 cm, unfilled bar) following afforestation by forest type. Figure 4.2b. The mean forest soil organic carbon concentration (SOC, 0-30 cm, filled bar, %) and non-forest site SOC (0-30 cm, unfilled bar) following afforestation by forest type. Figure 4.2c. The mean forest soil organic carbon density (SOCD, 0-30 cm, filled bar, Mg C ha^{-1}) and non-forest site SOCD (0-30 cm, unfilled bar) following afforestation by forest type. Figure 4.2d. The mean forest SOCD and forest floor carbon density (FFCD, filled bar, Mg C ha^{-1}) and non-forest site SOCD and forest floor carbon density (unfilled bar) following afforestation by forest type. Error bars represent standard deviation

4.4.2.2. Pre-afforestation land use

The 21 sites when arranged by pre-afforestation land use (Figure 4.3) were comprised of: 9 pasture sites; 9 rough grazing sites and 3 scrub sites. Pasture was defined as improved or unimproved grassland used for grazing animals. Rough grazing was defined as unimproved, poor grasslands often with species such as gorse and bracken and used for low density grazing animals. Scrub was defined as unplanted broadleaf forest that is often semi-natural. The SOCD of the forest sites were not significantly different to the non-forest SOCD, with each pre-afforestation land use declining after afforestation. The SOCD of the rough grazing sites decreased by a small amount following afforestation, 2.2% from the non forest site value of 117.9 ± 35.4 Mg C ha^{-1}, to the forest mean value of 115.3 ± 35.4 Mg C ha^{-1}. The pasture and scrub sites lost a similar amount of SOCD. The SOCD of the pasture sites decreased by 7.5% from the non-forest site value of 113.1 ± 21.6 Mg C ha^{-1} to the forest mean value 104.5 ± 13.7 Mg C ha^{-1}. The scrub sites SOCD decreased by 6.8%, from the non-forest site of 88.7 ± 40.2 Mg C ha^{-1} to the forest value of 82.6 ± 39.1 Mg C ha^{-1}.

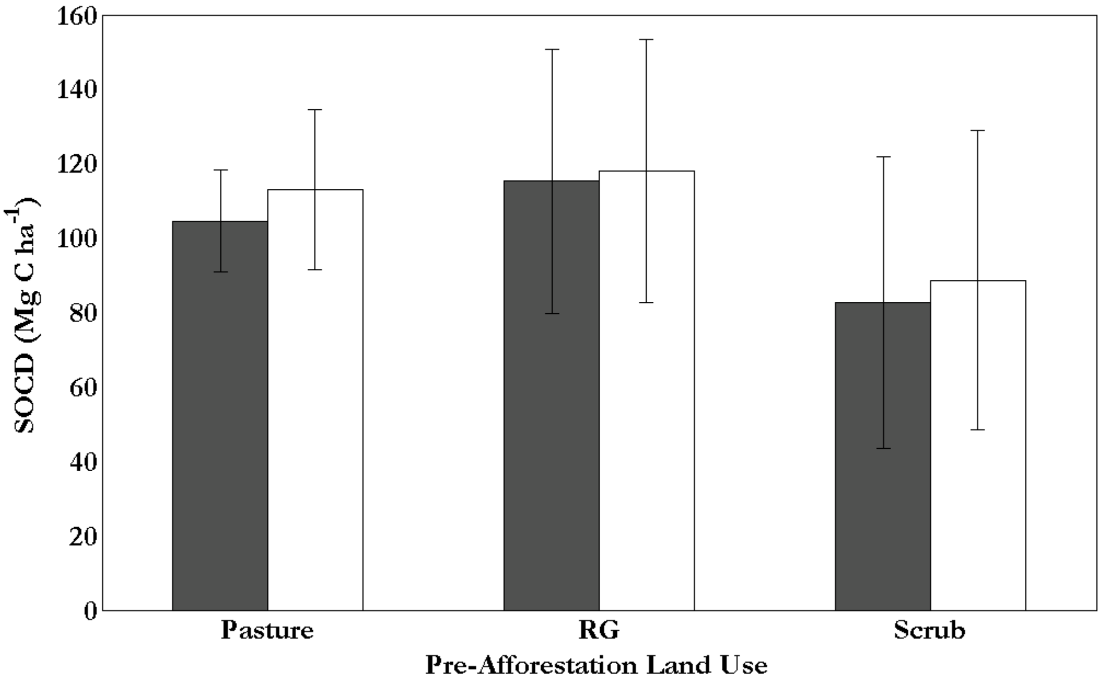

Figure 4.3. The mean forest soil organic carbon density (SOCD, filled bar, Mg C ha^{-1}) and non-forest site SOCD (unfilled bar) following afforestation by pre-afforestation land use. RG, rough grazing. Error bars represent standard deviation.

4.4.2.3. Soil texture

The 21 sites were partitioned by soil texture class (Figure 4.4) comprising of: 9 loam sites; 10 sandy loam sites; 1 clay loam site and 1 silt loam site. There were no significant differences between the SOCD of the forest site and non-forest following afforestation of the four soil textures. The SOCD of the two texture classes with multiple sites both decreased slightly following afforestation. The loam sites declined by 6.4% following afforestation from the non-forest site value of 113.4 ± 18.9 Mg C ha^{-1} to 106.1 ± 12.5 Mg C ha^{-1} at the non-forest site, while the sandy loam sites declined by 10.0% from 109.4 ± 42.1 Mg C ha^{-1} to 98.5 ± 35.8 Mg C ha^{-1} at the forest sites. The SOCD of the silt loam and clay loam sites both increased following afforestation. The silt loam sites increased by 3.7% from the non-forest site of 117.9 Mg C ha^{-1} to the forest sites 122.2 Mg C ha^{-1}, while the SOCD of the clay loam sites increased by 45.7% from the non-forest value of 112.7 Mg C ha^{-1} to the forest site of 164.3 Mg C ha^{-1}.

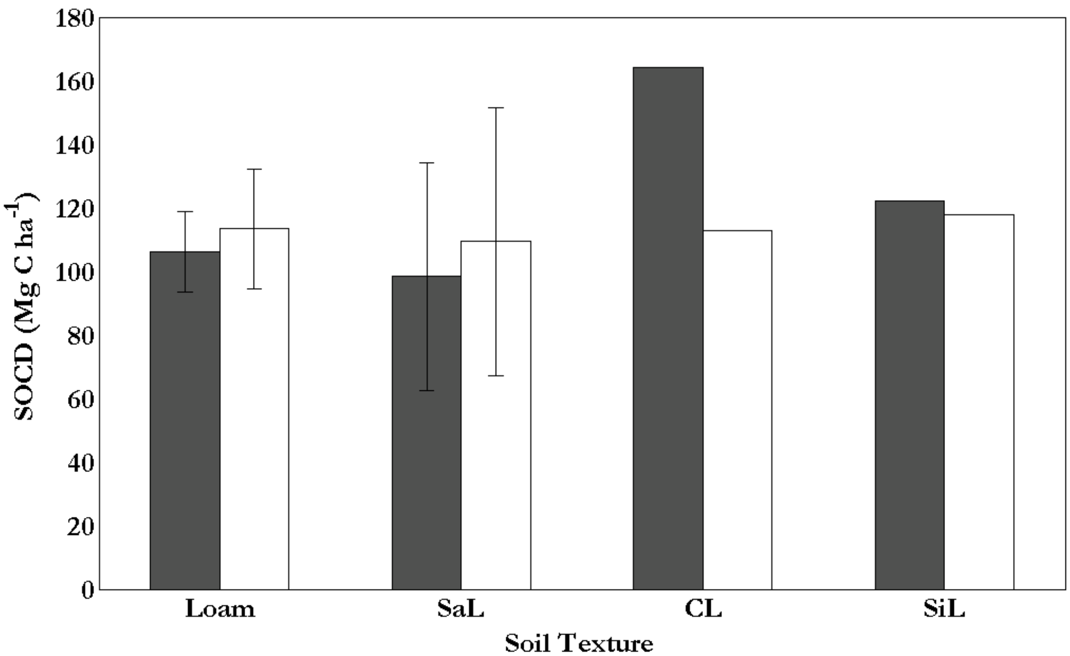

Figure 4.4. The mean forest soil organic carbon density (SOCD, filled bar, Mg C ha^{-1}) and non-forest site SOCD (unfilled bar) following afforestation by soil texture group. SaL, sandy loam; CL, clay loam; SiL, silt loam. Error bars represent standard deviation.

4.4.2.4. Soil pH

The soil of site CP4 was not measured for pH and so 20 of the 21 sites were analysed. The 20 sites were split into two pH groups: 7 sites with a pH <5; 13 sites with a pH between 5-7. There were no significant differences between the SOCDs of the forest site and non-forest site following afforestation of the two soil pH groups. The SOCD of sites with a soil pH <5 increased by 2.3% from the pre-afforestation mean value of 98.9 ± 39.8 Mg C ha^{-1} to the mean forest value of 101.2 ± 43.5 Mg C ha^{-1}, while those sites with a pH 5-7 lost soil SOCD by 5.0% from the paired site mean value of 107.8 ± 21.2 Mg C ha^{-1} to the mean forest site value of 113.5 ± 18.1 Mg C ha^{-1} (see Figure 4.5).

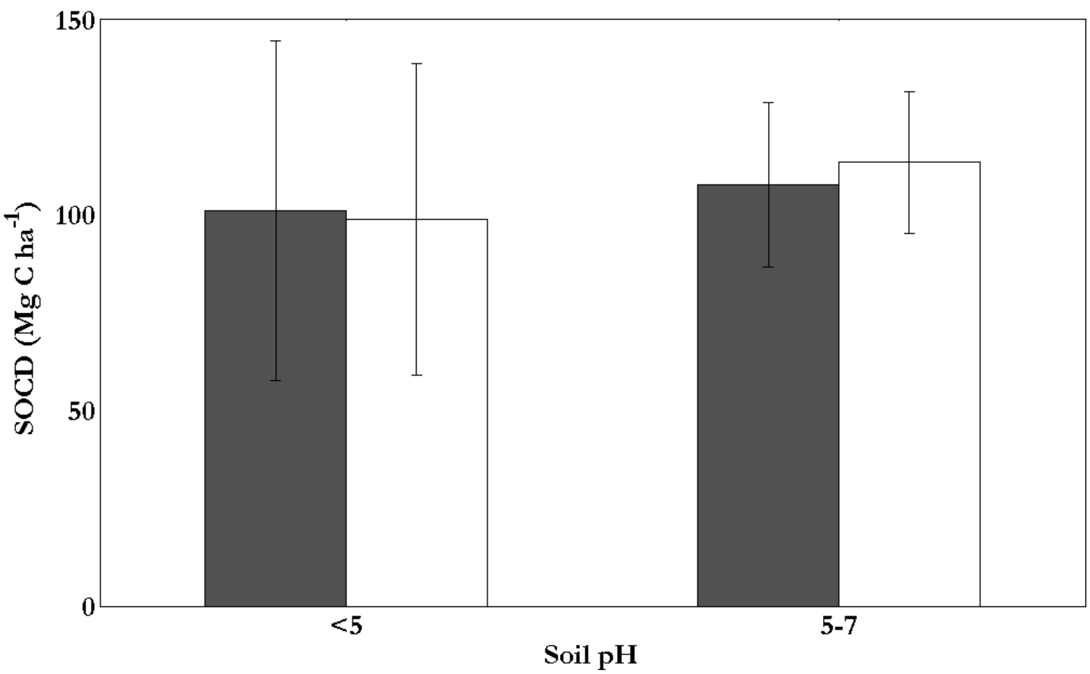

Figure 4.5. The mean forest soil organic carbon density (SOCD, filled bar, Mg C ha^{-1}) and non-forest site SOCD (unfilled bar) following afforestation by soil pH group, <5 and 5-7 pH. Error bars represent standard deviation.

4.4.2.5. Cultivation disturbance

The 21 sites were separated into two categories of cultivation disturbance prior to planting; 8 low-level disturbance sites and 13 medium/high-level disturbance sites. Low-level disturbance was defined as those sites that received no preparation or pit planting (planting of trees into a pit opened with a spade., no mechanical site preparation is used), was used while the medium/high-level disturbance sites were defined as those sites that were cultivated using machinery (agricultural ploughing, mounding, single mould board ploughing, double mould board ploughing and ripping).

There were no significant differences between the SOCDs of the forest and non-forest sites following afforestation of the two cultivation disturbance groups (Figure 4.6). The SOCD of the low-level disturbance recorded a loss of 11.2% from a pre-afforestation mean value of 95.7 ± 32.3 Mg C ha^{-1} to the forest mean value of 84.3 ± 23.2 Mg C ha^{-1}, while the SOCD of the medium/high-level disturbance sites recorded a small increase of 1.7% from the non-forest site of 121.5 ± 26.3 Mg C ha^{-1} to the forest value of 84.3 ± 23.2 Mg C ha^{-1}.

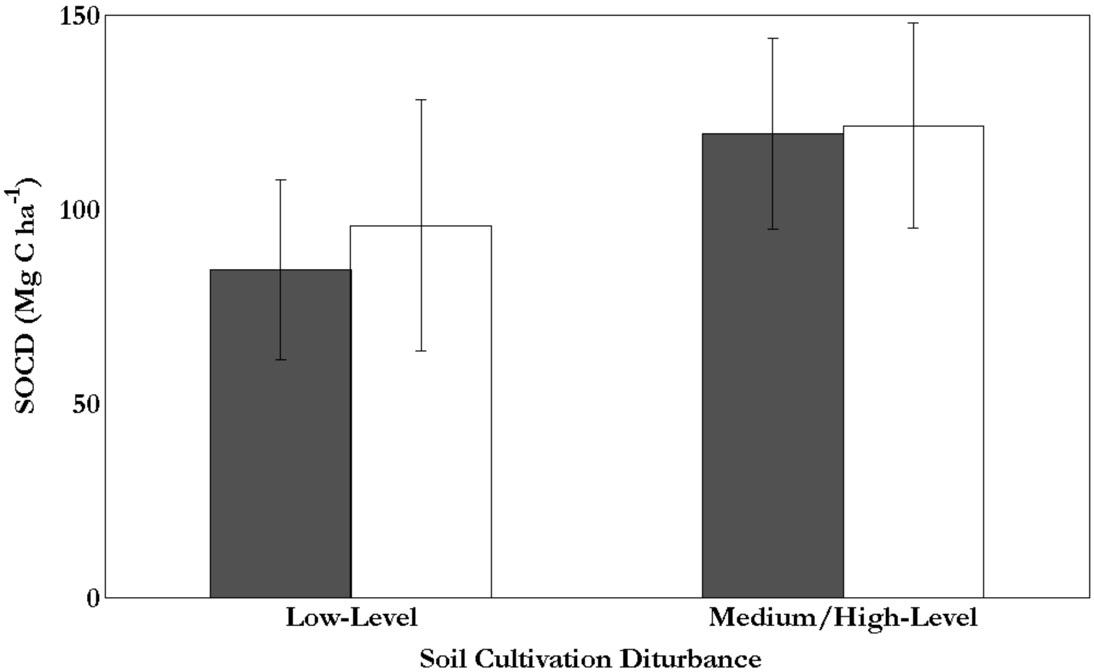

Figure 4.6. The mean forest soil organic carbon density (SOCD, 0-30 cm, filled bar, Mg C ha^{-1}) and non-forest site SOCD (0-30 cm, unfilled bar) following afforestation by level of soil cultivation disturbance. Error bars represent standard deviation.

4.4.2.6. Precipitation

The 21 sites when split by mean annual precipitation (Figure 4.7) were comprised of: 6 sites of 750-1000 mm; 11 sites of 1001-1250 mm, 2 sites of 1251-1500 mm, 1 site of 1501-2000 and 1 site of 2001-2500 mm. The mean annual precipitation of each site was taken from the Met Eireann 1960 to 1990 climate norms.

There were no significant differences between the bulk densities (0-30 cm) of the forest and non-forest sites for each precipitation group. The bulk density of the 750-1000 mm group decreased upon afforestation by 3.4% from the non-forest value of 1.06 ± 0.16 g cm^{-3} to the forest value of 1.02 ± 0.13 g cm^{-3} (see Figure 4.7a). The bulk densities of the 1001-1250 groups increased after afforestation by 11.4% from the non-forest value of 0.85 ± 0.18 g cm^{-3} to the forest value of 0.95 ± 0.14 g cm^{-3}. The bulk density of the 1251-1500 mm group increased by 15.5% from the non-forest value of 0.84 ± 0.05 g cm^{-3} to the forest value of 0.98 ± 0.39 g cm^{-3}. The bulk density of the 1501-2000 mm group decreased by 14.3% from the non-forest value of 0.53 g cm^{-3} to the forest value of 0.45 g cm^{-3} while the bulk density of the 2001-2500 mm groups increased by 6.7% from the non-forest value of 0.65 g cm^{-3} to the forest value of 0.69 g cm^{-3}.

There were no significant differences between the SOC (0-30 cm) of the forest and non-forest sites for each precipitation group (Figure 4.7b). The SOC of the 750-1000 mm sites decreased by 8.0% from the non-forest value of 4.0 ± 1.7%, to the forest value of 3.7 ± 1.1%. The SOC of the 1001-1250 mm sites decreased by 11.4% from the pre-afforestation level of 5.0 ± 1.9% to the forest value of 4.4 ± 1.6%. The 1251-1500 sites SOC decreased by 30.0% from the non-forest value of 4.7 ± 1.6% to the forest site value of 3.3 ± 0.01%. The SOC of the 1501-2000 mm increased by 93.0% from the non-forest value of 7.1% to the forest value of 13.7%, while the SOC of the 2001-2500 mm decreased by 19.3% from the mean non-forest value of 10.4% to the forest value of 8.4%.

The SOCD of the forest sites for the three precipitation levels were not significantly different to the non-forest SOCD (Figure 4.7c). The SOCD of each precipitation group declined, except for the 1500-2000 mm group which increased following afforestation. The SOCD of the 750-1000 mm sites decreased by 8.0% from the mean non-forest value of 113.9 ± 26.2 Mg C ha^{-1} to 104.8 ± 11.6 Mg C ha^{-1} at the non-forest site and the SOCD of the 1001-1250 mm sites decreased by 3.4% following afforestation from the non-forest site mean of 106.6 ± 33.0 Mg C ha^{-1} to the forest value of 103.0 ± 32.8 Mg C ha^{-1}. The SOCD of the 1251-1500 mm sites decreased by 22.4% from the mean non-forest value of 104.1 ± 25.3 Mg C ha^{-1} to 80.8 ± 8.3 Mg C ha^{-1} at the non-forest site while the SOCD of the 1501-2000 mm sites increased by 35.9% from the mean non-forest value of 109.5 Mg C ha^{-1} to 148.8 Mg C ha^{-1} at the non-forest site. The one site with a mean annual precipitation >2000 mm decreased by 9.9%, from the non-forest SOCD of 171.1 Mg C ha^{-1} to 154.1 Mg C ha^{-1} at the forest site.

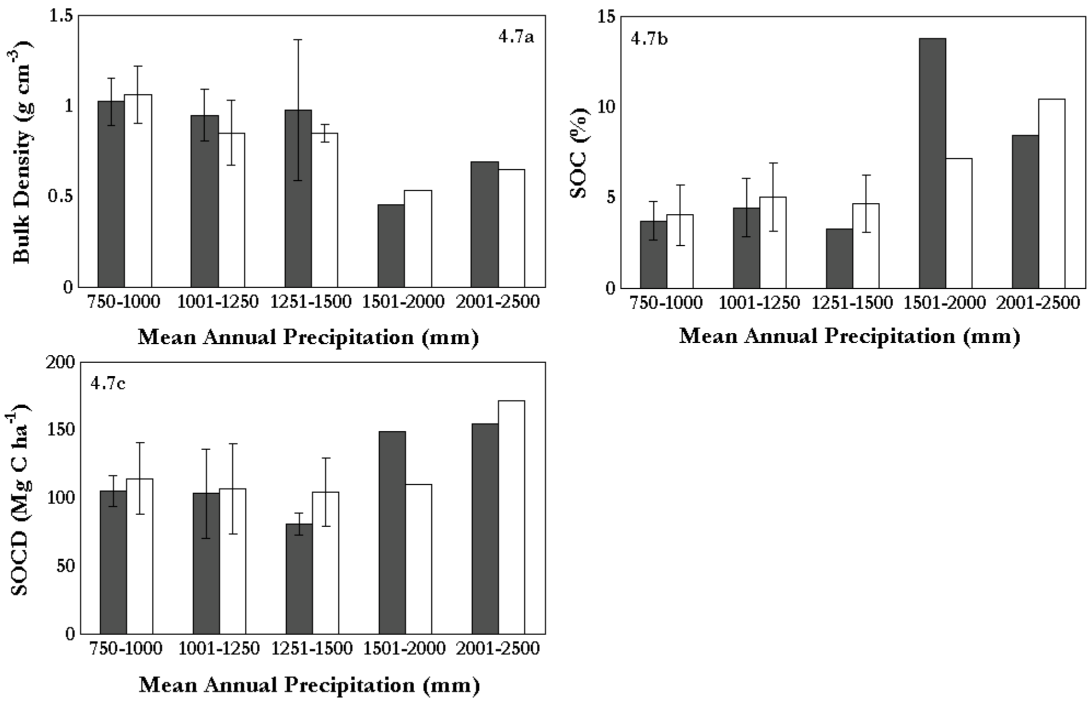

Figure 4.7a. The mean forest bulk density (0-30 cm, filled bar, g cm^{-3}) and non-forest site bulk density (0-30 cm, unfilled bar) following afforestation by mean annual precipitation (mm). Figure 4.7b. The mean forest soil organic carbon concentration (SOC, 0-30 cm, filled bar, %) and non-forest site SOC (0-30 cm, unfilled bar) following afforestation by mean annual precipitation (mm). Figure 4.7c. The mean forest soil organic carbon density (SOCD, 0-30 cm, filled bar, Mg C ha^{-1}) and non-forest site SOCD (0-30 cm, unfilled bar) following afforestation by mean annual precipitation (mm). Error bars represent standard deviation.

4.4.2.7. Age of forest

The 21 sites were arranged into four age groups: 5 sites aged 10-19 years, 5 sites aged 20-29 years, 4 sites aged 30-39 years, 7 sites aged >40 years. Statistically there were no significant differences between the forest site and non-forest site following afforestation of the four age groups. The SOCD of the 10-19 year group increased following afforestation by 1.5% from the non-forest value of 120.5 ± 32.5 Mg C ha^{-1} to the forest mean value of 122.3 ± 26.3 Mg C ha^{-1}. The SOCD of both the 20-29 and 30-39 year forests declined following afforestation. The 20-29 year SOCD decreased by 15.5% from the non-forest site value of 138.6 ± 25.9 Mg C ha^{-1} to the forest site value of 117.2 ± 28.7 Mg C ha^{-1}. The SOCD of the 30-39 year forests declined by 7.3% from the 110.9 ± 9.4 Mg C ha^{-1} of the non-forest site to 102.9 ± 6.9 Mg C ha^{-1} at the forest site. The >40 year group increased by a small 2.0% from the non-forest value of 86.5 ± 23.4 Mg C ha^{-1} to the forest value of 88.2 ± 26.3 Mg C ha^{-1} (see Figure 4.8).

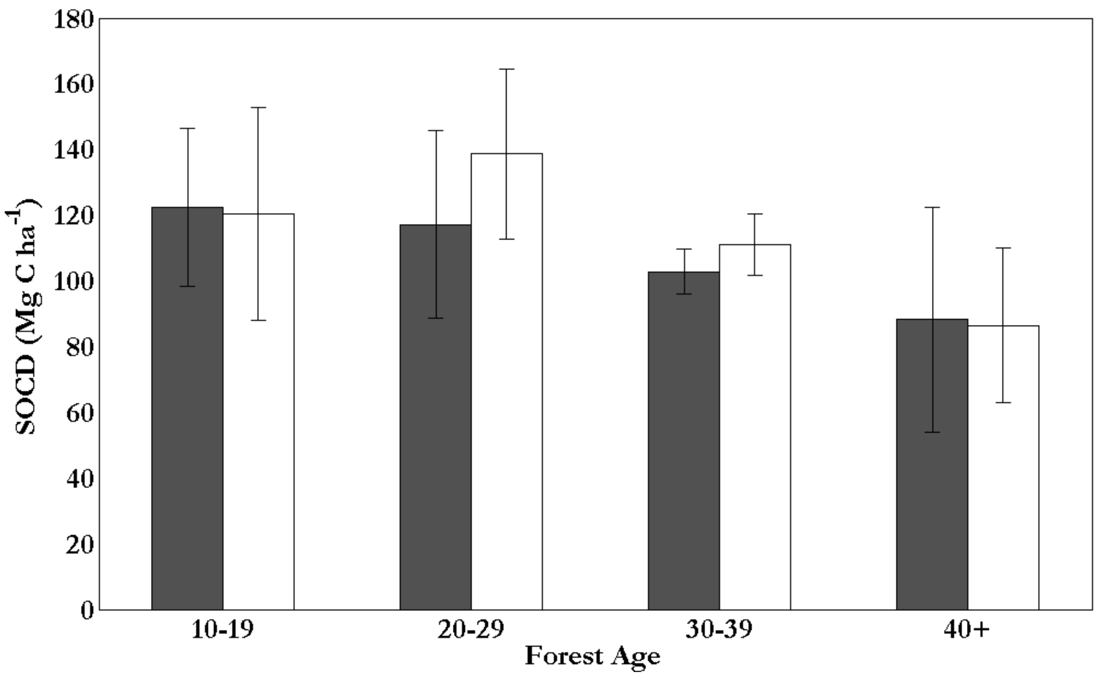

Figure 4.8. The mean forest soil organic carbon density of the forest (SOCD, filled bar, Mg C ha^{-1}) and non-forest site (unfilled bar) following afforestation by age group. Error bars represent standard deviation.

Table 4.2. The mean carbon density ± standard deviation (Mg C ha^{-1}) stored in each forest site and non-forest site. FWD, fine woody debris.

Site	Carbon Density (Mg C ha^{-1})								Change in Ecosystem (SOCD+FFCD)
	Litter + FWD		F/H		Soil (0-30 cm)		Ecosystem (SOCD+FFCD)		
	Forest	Non-Forest	Forest	Non-Forest	Forest	Non-Forest	Forest	Non-Forest	
BLBE1	1.6 ± 0.3	-	0.9 ± 0.2	-	98.6 ± 31.2	96.3 ± 2.2	101.1 ± 31.2	96.3 ± 2.2	4.8
BLBE2	1.0 ± 0.3	-	1.0 ± 0.1	-	111.2 ± 9.8	130.0 ± 14.2	113.1 ± 9.8	130.0 ± 14.2	-16.9
BLBE3	0.3 ± 0.5	-	0.2 ± 0.1	-	136.6 ± 18.6	107.2 ± 2.5	137.1 ± 18.6	107.2 ± 2.5	30
CBE1	4.5 ± 1.7	-	4.2 ± 2.2	-	95.6 ± 12.4	123.9 ±7.2	104.3 ± 12.7	123.9 ± 7.2	-19.6
CBE2	4.8 ± 0.9	-	9.3 ± 2.7	-	99.1 ± 9.6	118.7 ± 14.3	113.2 ± 10.0	118.7 ± 14.3	-5.5
CBE3	4.7 ± 3.2	1.6 ± 0.8	4.0 ± 1.5	2.6 ± 1.7	38.4 ± 2.2	43.1 ± 10.4	47.1 ± 4.1	47.4 ± 10.6	-0.3
CG1	1.3 ± 0.7	-	1.5 ± 0.5	-	122.2 ± 18.1	117.9 ± 14.0	125.0 ± 18.1	117.9 ± 14.0	7.1
CG2	2.9 ± 0.3	0.7 ± 0.2	3.5 ± 2.4	-	112.6 ± 16.0	103.5 ± 21.8	119.0 ± 16.2	104.2 ± 21.8	14.8
CG3	2.4 ± 0.8	-	11.7 ± 1.8	-	86.6 ± 28.9	122.0 ± 14.6	100.7 ± 29.0	122.0 ± 14.6	-21.3
CG4	1.7 ± 0.2	-	2.4 ± 0.7	-	164.3 ± 23.1	112.7 ± 19.6	168.4 ± 23.1	112.7 ± 19.6	55.6
CP1	2.3 ± 0.7	-	2.9 ± 0.6	-	103.0 ± 28.9	82.3 ± 20.3	108.2 ± 28.9	82.3 ± 20.3	25.9
CP2	4.5 ± 0.6	-	11.4 ± 4.4	-	148.8 ± 34.4	109.5 ± 16.1	164.7 ± 34.7	109.5 ± 16.1	55.2
CP3	5.6 ± 1.7	-	12.2 ± 9.4	-	66.7 ± 11.9	82.3 ± 49.8	84.5 ± 15.3	82.3 ± 49.8	2.2
CP4	6.1 ± 8.9	-	6.8 ± 3.9	-	116.8 ± 24.5	177.3 ± 69.3	129.8 ± 26.4	177.3 ± 69.3	-47.4
CBP1	1.4 ± 0.3	-	1.3 ± 0.5	-	74.9 ± 5.6	86.1 ± 11.2	77.6 ± 5.6	86.1 ± 11.2	-8.6
CBP2	2.2 ± 1.0	-	5.5 ± 3.1	-	107.1 ± 13.3	151.3 ± 11.0	114.9 ± 13.7	151.3 ± 11.0	-36.4
MBE1	0.8 ± 0.1	-	1.3 ± 0.3	-	101.4 ± 13.7	112.1 ± 18.8	103.4 ± 13.7	112.1 ± 18.8	-8.7
MBE2	2.6 ± 1.5	-	3.4 ± 1.3	-	88.3 ± 20.7	75.9 ± 6.4	94.4 ± 20.8	75.9 ± 6.4	18.5
MBE3	2.0 ± 0.6	-	2.8 ± 1.9	-	102.9 ± 6.1	102.2 ± 12.3	107.6 ± 6.4	102.2 ± 12.3	5.3
MG1	3.0 ± 0.9	2.4 ± 1.0	-	1.7 ± 0.9	96.9 ± 1.0	119.4 ± 8.6	99.8 ± 1.3	123.5 ± 8.7	-23.7
MG2	0.2 ± 0.2	-	-	-	154.1 ± 41.9	171.1 ± 52.1	154.3 ± 41.9	171.1 ± 52.1	-16.8
Mean	2.7 ± 1.7	-	4.1 ± 3.9	-	106.2 ± 29.2	111.7 ± 30.8	112.8 ± 28.2	112.1 ± 30.3	0.7

4.5. Discussion

We found that the tree species planted can influence the change in SOCD following afforestation, with broadleaf species sequestering more C to the soil than conifer species. However, it must be noted that these changes were not significant. Other studies have made the same conclusion. Laganière et al. (2010) and others (Guo and Gifford, 2002; Shi and Cui, 2010) found that the tree species planted plays a major role in the recovery of soil C, with broadleaf species sequestering significantly more C (25% increase on pre-afforestation levels) than pine forests and conifer forests when planted on agricultural land. These changes are due to how each tree species allocates biomass. Broadleaf forests have a larger and deeper root system (Strong and La Roi, 1983) than the shallow rooting conifer species (Jandl et al., 2007). The larger belowground biomass of broadleaves produces a larger SOM input to the soil than that of the conifer species, leading to an enhanced effect on SOC. However, this result is in contrast to the findings at two Irish chronosequences. Wellock et al. (2011a) in a chronosequence of ash forests in central Ireland reported an overall decrease of 21% of SOCD over the 47 years, while Black et al. (2009) in a chronosequence of Sitka spruce in eastern Ireland saw a large increase in SOCD from the pre-afforestation value of 97.2 to 137.3 Mg C ha^{-1} at the 16 year old site. In this study, we found that when the forest floor carbon density was added to the soil organic carbon density, the conifer forests switched from a source of C to a sink, which has been reported in the literature (Paul et al., 2002; Laganière et al., 2010). The litter of conifer species decomposes more slowly than broadleaf forests (Vesterdal and Raulund-Rasmussen, 1998) and so it is critical to measure the forest floor carbon density when evaluating soil C stock changes as the full sequestration potential of conifer forests following afforestation is not identified without the forest floor contribution (Laganière et al., 2010).

If the volume based method was used to analyse the SOCD, the conifers would have switched from losing C to sequestering C as the increase in bulk density is larger than the reduction in SOC concentration. While the decline in bulk densities of the soils in broadleaf forests are larger than the increase in SOC, and so would lead to a decline in SOCD within broadleaf forests following afforestation. The volume based method does not take into account that bulk densities can vary following land use change (i.e. afforestation), and can lead to an over- or underestimation of the SOCD. This highlights the importance of using the equivalent soil mass (ESM) corrections proposed in Ellert and Bettany (1995) and Lee et al. (2009), for assessments of SOCD change following changes of land use or management.

The literature shows that afforestation of pastures and rough grazing sites produces either a negligible effect or a reduction in SOC following afforestation which was also observed in this study. Guo and Gifford (2002) noted a 10% reduction in soil C following conversion from pasture sites to plantation forest, with Paul et al. (2002) noting a similar result. However, Laganière et al. (2010) recorded a small increase in soil C of 3% for pastures and a 10% increase in afforested grasslands. A large proportion of the tree biomass is concentrated in the trunk, aboveground, while herbaceous plants allocate most of their biomass belowground into the root system (Cerri et al., 1991; Kuzyakov and Domanski, 2000; Bolinder et al., 2002). Tree roots also live longer than those of herbaceous plants, and so the annual turnover of organic matter into the soil from dying grass roots is larger than that from trees (Guo and Gifford, 2002). As in this study Guo and Gifford (2002) recorded a loss of 13% from forest (scrub) sites to plantation forestry which was attributed to soil disturbance from deforestation and subsequent afforestation. Guo and Gifford (2002) also noted that the soil C of the plantation sites was restored to pre-afforestation values after 40 years. The forest sites presented in this study that were previously scrub have a mean age of 26 years, and so the loss of soil C may be due to the forest sites not being old enough to have restored the soil C stocks to previous levels.

Afforestation of clay loam and silty loam sites in this study increased the SOCD, which has been noted in the literature when high clay content soils were afforested (Paul et al., 2002; Laganière et al., 2010). Fine particles (clay and silt) form stable organo-mineral complexes with organic compounds and these complexes provide physical protection against microbial oxidation for SOM and contribute to the stability of SOC (von Lutzow et al., 2006). However, only one of the 21 sites in this study was classified as either clay loam or silt loam so we are unable to determine the true effect of soil texture on SOCD in Ireland.

Low pH can retard the activities of soil microorganisms that decompose SOM and lead to the preservation of SOM inputs to the soil (Motavalli et al., 1995; van Bergen et al., 1998), which was shown in this study. Our results are in contrast to Laganière et al. (2010), that observed an increase in SOC following afforestation with increasing pH, but also found it to be the weakest factor in determining soil C stock changes. Low pH can impair tree growth and thus its C inputs to the soil and may explain the results of Laganière et al. (2010). The results of Laganière et al.

(2010) show that the effects of pH on soil C stock changes with afforestation are still uncertain, and further work is required determine the true effect.

It has been shown in the literature that soil disturbance from cultivation creates a loss of soil C due to enhanced microbial decomposition of SOM, and the greater the soil disturbance the more C that is lost to the atmosphere (Johnson, 1992; Örlander et al., 1996; Schmidt et al., 1996; Mallik and Hu, 1997; Bock and Van Rees, 2002); however, this was not observed in our study. Laganière et al. (2010) found that low pre-planting disturbance sequestered a significantly higher amount of soil C than the high pre-planting disturbance sites. It is unclear why we observed the opposite trend. The mean ages of the low disturbance sites and high disturbance sites are 49 and 25 years, respectively. It would be assumed that the medium/high-level disturbance sites would see a greater loss of C following afforestation due to the greater effects of soil disturbance and also as young forests they would also have a much smaller organic matter input to the soil; however, this was not observed.

We found that the SOC concentration and soil organic carbon density (SOCD) of both the forest site and non-forest site increases when annual precipitation was >1500 mm, as high precipitation is known to suppress decomposition of organic matter. However, we found no decline in the SOCD following afforestation in the high rainfall sites; even though it has been reported that sites with higher rainfall lose more SOC than drier sites (Guo and Gifford, 2002; Jackson et al., 2002; Kirschbaum et al., 2008). Guo and Gifford (2002) found that there was little change for sites with an annual precipitation <1200 mm while sites with precipitation >1200 mm saw a significant decline in soil C stocks following afforestation. Kirschbaum et al. (2008) suggest that in high rainfall areas, larger amounts of nitrogen were leached from the soil, leading to a decline in the soil C stocks, while the drier sites had little leaching of C and so did not lose as much C from the soil. However, in China, Shi and Cui (2010) noted that all sites saw an increase in soil C following afforestation despite the precipitation level. This highlights the variation in findings and suggests the need for a more detailed examination of precipitation levels in soil C change following afforestation.

Studies have shown that following afforestation, there is a decrease in SOCD for a number of years before it switches from a source of C to a sink, potentially matching or surpassing the initial pre-afforestation SOCD (Turner and Lambert, 2000; Davis and Condron, 2002; Paul et al., 2002; Wang et al., 2006; Huang et al., 2007; Ritter, 2007; Hu et al., 2008; Laganière et al., 2010; Mao et al., 2010). The age effects of SOCD on this study suggests that it may take 40 years or more for forest soils to sequester SOCD to pre-afforestation levels. It was surprising that in the age group 10-19 years there was no loss in SOCD following afforestation as is widely reported in the literature. We found appreciable losses in SOCD following afforestation in the age groups 10-29 and 30-39.

It is current Irish government policy to increase forest cover from its current 10% (NFI, 2007a) to 17% by 2030 (Department of Agriculture, 1996). This provides an opportunity to establish a large number of resampling studies, covering a number of differing forest variables, i.e. forest type, pre-afforestation land use etc. The resampling studies should be identified prior to cultivation, so that the SOCD of the pre-afforestation land use can be measured to further investigate the effects of site cultivation and stand establishment on SOCD. Resampling studies are preferable to paired plot studies as they measure change over time, while the paired plot method only measures one point in time, and may overestimate SOCD (Laganière et al., 2010).

4.6. Conclusion

We found in this study that afforestation of Irish soils (0-30 cm) resulted in an increased soil organic carbon density (SOCD) for broadleaf afforestation but a decrease in SOCD for mixed and conifer plantations. However, when the additional forest floor carbon density was included, the conifer sites had a higher total C than their adjacent non forested sites, while the mixed forests remained with a lower total C than their adjacent non forest sites. We suggest that all future assessments should include measurements of the forest floor carbon density to analyse the full C sequestration potential of a forest stand. The SOCD of sites prepared by a medium/high-level disturbance cultivation method declined less than that of the low-level disturbance, highlighting the need for further research on the effects of SOCD change with differing soil cultivation methods. Afforestation of rough grazing sites recorded the smallest loss of SOCD among the different land uses, possibly due to the lower productivity of those sites compared to the pasture and rough grazing sites. Low pH soils resulted in increased SOCD. This suggests that the planting of broadleaf species and establishment of forests on rough grazing sites or low pH soils can maximise the sequestration potential of C into soil in Ireland. However, the number of sites and large variability presented in this study leads to uncertainty in estimating the impact of each variable on SOCD changes, and so more sampling sites are required to better estimate the importance of each factor in determining the impact of afforestation on SOCD.

The methodology used to assess the change in SOCD can impact on the possible findings. The use of the volume based method rather than the equivalent soil mass method can lead to an erroneous assessment of the impact of forest type on SOCD.

The large afforestation set to take place in Ireland within the next decade represents a unique opportunity to measure the impact of afforestation on SOCD. We recommend that resampling studies should be established prior to afforestation on a large number of sites across Ireland, and periodically resampled to gain a much greater insight into changes in the soil, forest floor and biomass C following afforestation.

4.7. Acknowledgments

This project "Soil Carbon Stock Changes and Greenhouse Gas Flux in Irish Forests (ForestC)" is funded by the Irish Council for Forest Research and Development (COFORD) as part of the Department of Agriculture Fisheries and Food (DAAF). We thank all the land owners for their help during the site sampling process.

5. Soil organic carbon stocks of afforested peatlands in Ireland

Michael L. Wellock[1], Brian Reidy[2], Christina M. LaPerle[1], Thomas Bolger[2] & Gerard Kiely[1]

[1] *Centre for Hydrology, Micrometeorology and Climate Change, Department of Civil and Environmental Engineering, University College Cork, Cork, Ireland.*
[2] *School of Biology and Environmental Science, University College Dublin, Belfield, Dublin 4, Ireland.*

Submitted to Forestry on January 20th 2011. Has been accepted for publication pending minor corrections.

Keywords: Soil Organic Carbon; SOC; Peatlands; Bulk Density; Peat Depth; Carbon Density.

5.1. Abstract

This paper assessed the soil organic carbon (SOC), bulk density, depth and soil organic carbon density (SOCD) of 24 afforested peatlands. We found that the peat bulk density does not increase with depth, as has been previously understood in the literature. The depths of each peat type were found to vary widely with means of 192 ± 100, 145 ± 130 and 127 ± 100 cm for raised bogs (RB), high level blanket bog (HLB) and low level blanket bog (LLB), respectively. Based on the full surveyed depth, we estimated soil organic carbon densities of: 1160 ± 520 Mg C ha^{-1} for RB peat; 775 ± 590 Mg C ha^{-1} for HLB peat; and 705 ± 420 Mg C ha^{-1} for LLB peat. We found peat depth and peat type to be significant predictors of peat SOCD and present pedo-transfer functions for SOCD based on these predictors that will help to improve future peatland SOCD estimates. We suggest that due to the similarities between the SOCDs of the high level and low level blanket peats, they can be analysed as one group for accounting purposes and future SOCD estimates.

5.2. Introduction

Although peatlands cover only approximately 3% of the Earth's land surface, they provide a significant carbon (C) store, with boreal and subarctic peatlands estimated at 270 and 370 Pg of carbon (C) (Turunen et al., 2002). This is equivalent to 34-46% of the total carbon (as carbon dioxide, CO_2) in the atmosphere (IPCC, 2007). An estimated eighty percent of peatlands are in the northern hemisphere, particularly in Russia, Canada and the USA (Limpens et al., 2008) with smaller areas in Ireland and other European nations. Approximately 20.6% (Connolly and Holden, 2009) of the Irish land area is peatland, containing an estimated soil carbon store of between 53 - 62% of the national soil carbon stocks (Tomlinson, 2005; Eaton et al., 2008).

Peatlands have long been considered carbon sink ecosystems. However, with the onset of climate change, peatlands may become carbon source ecosystems with the potential to lose carbon (from their large carbon stocks) either as trace gases such as carbon dioxide (CO_2) and methane (CH_4) or fluvial dissolved organic carbon (DOC) (Koehler et al., 2010). One direct feedback to rising greenhouse gases in an anthropogenically warmed environment is the release of organic carbon stored in vegetation and soils. To assess this potential release of carbon and its effect on climate change, accurate estimates of the quantity and distribution of soil organic carbon density (SOCD) are required (Burnham and Sletten, 2010). The soil organic carbon density is estimated as the product of bulk density (g cm^{-3}) and soil organic carbon concentration (SOC, %) over the full peat depth. Peatland carbon stock studies in Scotland (Chapman et al., 2009) and in Ireland (Tomlinson, 2005) have used models of peat based on either assumed values or a small sample size for peat bulk density and depth. Tomlinson (2005), Limpens et al. (2008) and Chapman et al. *(*2009) have identified the need for extensive data collection of bulk density and peat depth to improve the current estimates of soil organic carbon density. The spatial extent of peat (but not its depth) in Ireland has been examined by Connolly et al. (2007) and Hammond (1981), but there have been few studies that quantify the physical properties (e.g. bulk density, peat depth and SOC) of the major peat types (e.g. raised bog, high level and low level blanket bog) (Tomlinson and Davidson, 2000; Tomlinson, 2005; Eaton et al., 2008; Kiely et al., 2010).

Ground penetrating radar (GPR) is becoming a useful tool in assessing the depth of peatlands (Holden et al., 2002; Rosa et al., 2009). However, the most commonly used method for determining peat depth is probing with connecting metal rods (Lindsay, 2010). Probing provides a cheap, quick estimate of depth, but is limited in that it only gives the depth of the specific location, and the peat depth can differ substantially only a few metres away (Lindsay, 2010). In assessing national and global peat stocks, studies often assume a mean depth value for all peat, or mean values for different peatland types or regions. Tomlinson (2005) in his assessment of soil C stocks, separated peat depth estimates by peat type, location and its current and past use by man. Intact midland raised bogs, western raised bogs, northwestern, high level and low level blanket bogs were assigned depths of 750, 400, 300, 120 and 300 cm, respectively. For man-modified peatlands, i.e. afforested peatland, a loss of 66% of the peat depth was assumed for all peatland types except for high level blanket peat which assumed a 50% loss. In estimating the carbon stock of peatlands, both within Ireland and globally, the lack of data on the depth of peatlands limits the accuracy of the stock estimate.

Soil organic carbon (SOC, %) is the most studied property of peat, with most values reported to be around 50%. Tomlinson and Davidson (2000) found a mean SOC value of 51.1% for Northern Irish non-forested raised bogs, while Chapman et al. (2009) report values ranging from 50.6 to 54.6 % for Scottish blanket bogs.

There is very little data on peat bulk density for Ireland and Britain, especially below the 50 cm depth (Lindsay, 2010). Tomlinson and Davidson (2000) observed a mean bulk density of 0.069 g cm^{-3} for non-forested raised peats in Northern Ireland while Lewis et al. (2011) observed a mean value of 0.070 g cm^{-3} for a high level blanket bog in County Kerry, Ireland. Some authors have noted that the bulk density of peat increases with depth (Clymo, 1978; Howard et al., 1994; Milne and Brown, 1997; Cruickshank et al., 1998). However, a number of recent studies have shown that this may not be the case. Recent studies have shown either no consistent change in bulk density with depth (Tomlinson and Davidson, 2000; Lewis et al., 2011) or a slight increase with depth (Weiss et al., 2002).

Over recent centuries, Ireland has been denuded of its native forests, so much so that at the beginning of the twentieth century the national forest stock was estimated at only 1% of the total land cover in the Republic of Ireland (Pilcher and Mac an tSaoir, 1995; Eaton et al., 2008). Since the mid twentieth century, it has been Irish government policy to increase forest cover and by 2007 the national forest area had risen to 10% of the total land area (NFI, 2007a) with a projected increase to 17% by 2030 (Department of Agriculture, 1996).

Large areas of peatland in the boreal and temperate zones have been commercially forested, the majority in the Nordic countries and the former Soviet Union, while large areas have also been afforested in Ireland and the UK (Byrne and Farrell, 2005; Vasander and Kettunen, 2006). In the past, afforestation of peatlands was considered an attractive option for Ireland as peatlands were considered as marginal lands unsuitable for agriculture (Renou and Farrell, 2005; Byrne and Milne, 2006). This has resulted in peat soils being the largest soil type of Irish forests; with 42.1% of the total Irish forest cover (NFI, 2007a) on peat soils. The main forest species planted on peatlands were Sitka spruce (*Picea sitchensis* (Bong.) Carr.) and lodgepole pine (*Pinus contorta* Dougl.) in large monocultural, non-native stands (Byrne and Farrell, 2005). Afforestation of peatlands began in the 1950s and while still continuing into the 21st century, the rate of peatland afforestation has decreased from 56% of total annual afforestation in 1990 to 29% in 2003 (Black et al., 2009). This decrease is due to the poor economic return of forested peatland as well as the growing awareness that peatlands should be conserved not only because of the unique biodiversity of their landscapes (Foss et al., 2001) which contain significant amounts of carbon but also because these ecosystems are currently known to be carbon sinks (Sottocornola and Kiely, 2010).

Afforestation can impact the carbon balance of a peatland due to increased soil aeration which follows from lowering the water table through drainage and increased evapotranspiration. This leads to growth of microbial aerobic decomposers (Chmielewski, 1991; Byrne and Farrell, 2005) which enhance the rate of organic matter decomposition and loss of carbon as CO_2 (Lieffers, 1988; Bridgham et al., 1991) and can lead to a decrease in carbon stocks (Braekke and Finér, 1991; Sakovets and Germanova, 1992). However, there may also be an increase in the vegetative input of carbon to the peat through roots, litterfall and forest harvest residues (Anderson et al., 1992; Minkkinen and Laine, 1998; Reynolds, 2007). Such uncertainties highlight the need to sample peat properties to assess the current carbon stocks of afforested peatlands.

There are two major types of peatland in Ireland: fens and bogs. Bogs are forested more frequently with 40.7% of the total national forest cover occurring on them, while fens only represent 1% of the total forest cover (NFI, 2007a). Bogs are ombrotrophic ecosystems, taking their water supply from the mineral-poor rainwater and are of two types: blanket bogs and raised bogs. Globally, blanket bogs are a small part of the total peatland area, accounting for ca. 3% of the global peatland area (Foss et al., 2001). However, blanket bogs are an important form of peatland landscape in Ireland comprising approximately 18% of the total peat area (Hammond, 1981). Blanket bogs are further classified into two types: low level blanket bogs (LLB), have *Schoenus nigricans* as a large contributor to the vegetative cover while high level blanket bogs (HLB) do not (Hammond, 1981). Blanket bogs are located predominantly along the western seaboard and at higher altitudes and represent 0.196 Mha or 31.5% of the total forest area within Ireland (NFI, 2007a). Raised bogs (RB) are formed in areas with a high watertable due to impermeable subsoil, such as in hollows, lake basins and river valleys (Gardiner and Radford, 1980). Raised bogs are predominantly found in the midlands of Ireland and represent 0.066 Mha or 10.7% of the total national forest area (NFI, 2007a). In this study, peat was classified as having a depth greater than 30 cm (excluding the thickness of the plant/litter layer) and organic matter content greater than 30% (Hammond, 1981).

To enable improved estimates of carbon stocks in afforested peatlands, this study was designed: (1) to quantify the bulk density, SOC and depth of peatland soils of Irish forests on raised and (high level and low level) blanket bogs; (2) to estimate the current soil organic carbon density of Irish afforested peatland soils; and (3) to determine suitable pedo-transfer type functions for soil organic carbon density.

5.3. Materials and methods

5.3.1. Site selection and description

In 2007, the Irish Forest Service (IFS) produced a national forest inventory (NFI) with detailed field surveys of forest plots (NFI, 2007b). The NFI surveyed 1,742 forest sites that were selected from a randomised systematic grid sample design. After a pilot study in Co. Wexford, a grid density of 2 x 2 km grid was placed over the total land base of Ireland (6,976,100 ha). The NFI collected data on forest biology and geography, including forest type, age and soil type. From this NFI database, we selected a sub-set of afforested raised bog, high level blanket bog and low level blanket bog sites with forest age greater than 15 years and that were accessible by foot for sampling purposes. Fen bogs were excluded from our sub-set as they represent a small portion (1%) of the forested land cover (NFI, 2007a). We partitioned our sub-set into 3 categories: conifer forested raised bog; conifer forested high level blanket bog; and conifer forested low level blanket bog. Twenty-four sites were randomly selected from the 315 sites of the above sub-set representing the 3 forested categories and with an age range of 18 to 45 years. This resulted in the following distribution of sites among groups: 11 conifer forested raised bog sites; 6 conifer forested high level blanket bog sites; and 7 conifer forested low level blanket bog sites. Details and locations of the sites are shown in Table 5.1 and Figure 5.1.

Table 5.1. Characteristics of each site. RB (raised bog); HLB (high level blanket bog); LLB (low level blanket bog).

Site ID	Elevation (m)	Slope (°)	Tree Species	Forest Age (years)	Tree DBH	Georeference Site
RB1	48	2	*Pinus contorta*	31	301-400	53° 18' N 8° 32' S
RB2	50	1	*Picea sitchensis*	18	141-200	53° 26' N 8° 41' S
RB3	127	2	*Picea sitchensis*	25	71-140	52° 52' N 7° 42' S
RB4	114	1	*Pinus silvestris*	31	301-400	52° 4' N 9° 8' S
RB5	112	9	*Picea sitchensis*	31	71-140	54° 16' N 8° 14' S
RB6	53	3	*Pinus contorta*	40	301-400	53° 27' N 8° 16' S
RB7	68	2	*Picea sitchensis*	24	201-300	53° 37' N 8° 19' S
RB8	84	2	*Picea sitchensis*	20	71-140	53° 29' N 8° 34' S
RB9	85	1	*Picea sitchensis*	20	201-300	53° 25' N 7° 13' S
RB10	81	1	*Picea abies*	41	141-200	53° 28' N 7° 7' S
RB11	94	2	*Picea sitchensis*	30	71-140	53° 18' N 7° 25' S
HLB1	219	4	*Picea sitchensis*	18	31-70	54° 53' N 8° 0' S
HLB2	276	2	*Picea sitchensis*	39	301-400	52° 20' N 9° 28' S
HLB3	270	6	*Picea sitchensis*	41	141-200	51° 49' N 9° 23' S
HLB4	214	3	*Picea sitchensis*	18	31-70	52° 53' N 6° 29' S
HLB5	142	1	*Pinus contorta*	21	71-140	54° 13' N 9° 33' S
HLB6	286	3	*Picea sitchensis*	35	141-200	54° 19' N 8° 7' S
LLB1	75	2	*Picea sitchensis*	24	31-70	53° 18' N 9° 13' S
LLB2	56	0	*Picea sitchensis*	18	31-70	52° 11' N 10° 18' S
LLB3	56	1	*Picea sitchensis*	35	301-400	51° 53' N 9° 47' S
LLB4	121	0	*Picea sitchensis*	21	71-140	52° 46' N 9° 7' S
LLB5	50	0	*Pinus contorta*	25	201-300	53° 44' N 9° 49' S
LLB6	93	1	*Picea abies*	45	301-400	53° 19' N 6° 51' S
LLB7	136	3	*Picea sitchensis*	20	71-140	54° 11' N 9° 35' S

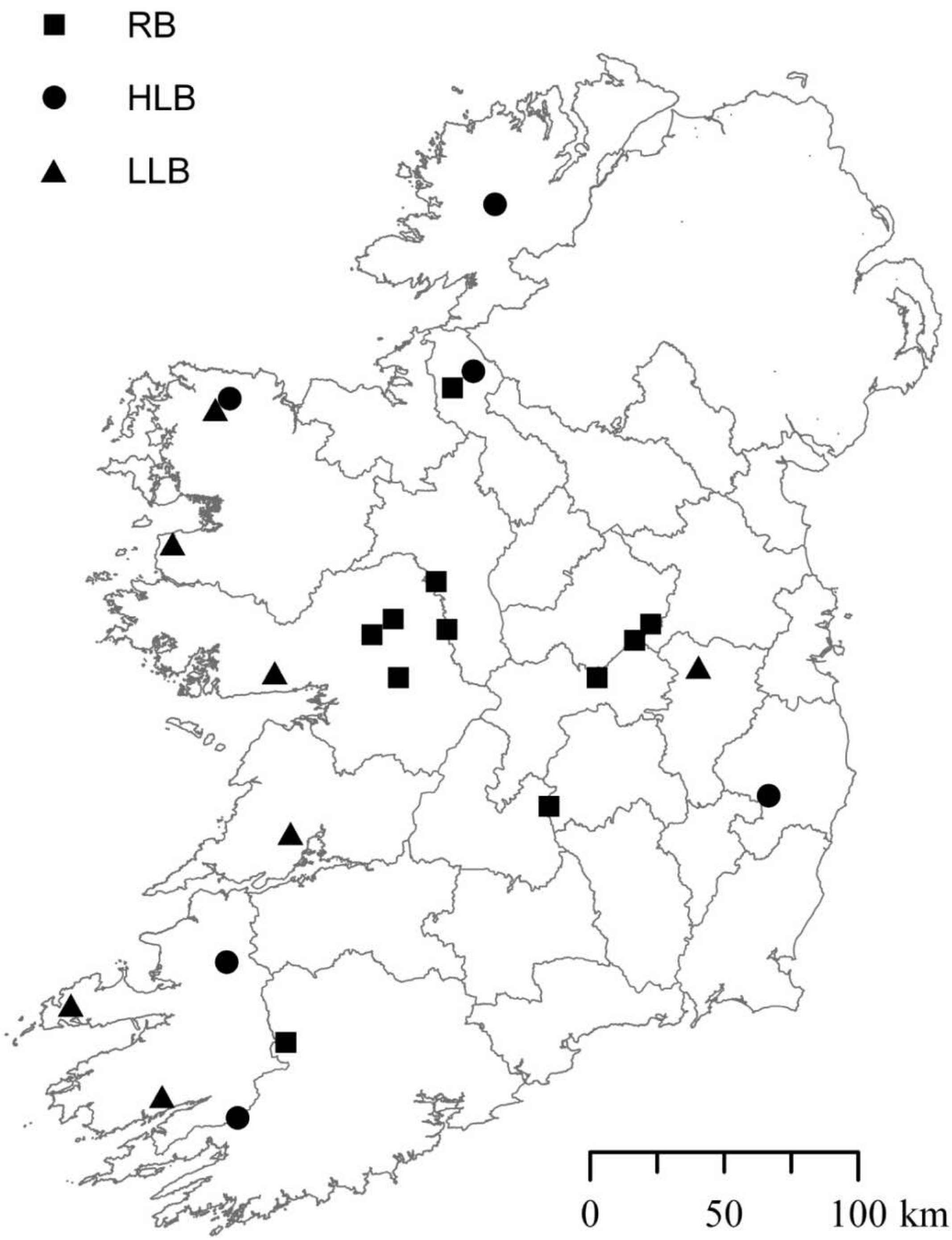

Figure 5.1. Locations of all 24 peatland sites within Ireland arranged by type. RB, raised bog; HLB, high level blanket bog; LLB, low level blanket bog.

5.3.2. Sampling methodology

Each NFI site was located using GPS (GPS 60 Garmin, USA). At each site a 20 m by 20 m square plot was set out, and then partitioned into four 10 m by 10 m quadrants. Within each quadrant, 4 points were randomly selected and the soil sampled using a Russian peat corer (Eijkenkamp Agrisearch Equipment BV, Netherlands) with sample length 50 cm, diameter 5.2 cm and a volume of 500 cm^3 for bulk density ($g\ cm^{-3}$) and SOC (%). The peat was sampled over its full depth from the peat surface vertically down the profile in increments of 50 cm. A second peat sample (at depth 50 to 100 cm) was taken at a point 10 cm west of the first sampling point to avoid the effects of compaction from the previously extracted sample and a third sample (100 to 150 cm) was taken 10 cm away from the second sampling point, while the fourth (150 to 200 cm) was from the original sampling point and so on. When the bottom of the peat was reached, indicated by the presence of a mineral layer or the presence of impenetrable rock, the final depth was recorded and the sample taken. If a sample was seen to contain a peat pipe or cavity, its depth was recorded with the sample ID. At sites RB1-5, HLB1-4 and LLB1-5, around each sampling point, 4 additional points were used to determine the peat depth using the Russian peat corer. Each point was located 50 cm north, south, east or west of the initial sampling point. This was repeated for each of the 4 random points within the 4 quadrants of the site.

There were limitations to sampling using the Russian peat corer in the top layer due to the inability of the corer to cut through the roots of the forest floor vegetation, usually most prominent in the top 20 cm. To compensate for this, a random point was chosen within each 10 m by 10 m quadrant where a hole was dug and samples were taken using stainless steel bulk density rings (Eijkenkamp Agrisearch Equipment BV, Netherlands) of 8 cm diameter by 5 cm height. Samples were taken with the rings to replace those that the Russian peat corer could not extract reliably. The Russian corer could not reliably sample 0–30 cm, and at each site we used the rings to sample at depths: 0-5 cm; 5-10 cm; 15-20 cm; and 25-30 cm (5 cm gaps were placed between samples after 10 cm to avoid the effects of compaction from previous samples).

To summarise, at each of the 24 sites we obtained: (1) 16 peat profiles using the Russian peat corer for bulk density and SOC laboratory measurements; (2) four surface profiles (typically 0-30 cm) using stainless steel rings for bulk density and SOC; and (3) 80 individual estimates of peat depth, using the Russian auger at sites RB1-5, HLB1-4, LLB1-5, and 16 individual estimates of peat depth at sites RB6-11, HLB5-6 and LLB6-7.

All samples were stored at 4 °C before being dried at 55 °C until a constant dry weight was achieved. The samples were then bulked by volume for each depth within each quadrant. The bulked samples were ground to a fine powder and the SOC (%) determined by combustion in a C/N analyzer (Elementar – Vario Max CN). The bulk density samples were sieved to 2 mm and were determined as the dry mass per fresh volume (g cm^{-3}). The bulk density was estimated using equation (5.1):

$$\rho_d = \frac{MA}{SV - CFV} \tag{5.1}$$

where ρ_d = bulk density (g cm^{-3}); MA = mass of dry sample <2 mm (g);
SV = sampler volume (cm^3); CFV = >2 mm coarse fraction volume (cm^3)

The soil organic carbon density of each peat layer was estimated using equation (5.2):

$$C_{i,vol} = \rho_{di} \times conc_i \times h \times 10^2 \tag{5.2}$$

where $C_{i,vol}$ = soil organic carbon density, volume-based method (Mg C ha^{-1}); $conc_i$ = soil organic carbon (%); h = Depth of Peat $\geq$ 30 cm.

The soil organic carbon density over the entire peat depth is calculated using equation (5.3).

$$C_{tot} = \sum C_{i,vol} \tag{5.3}$$

where C_{tot} = total soil organic carbon density

5.3.3. Statistical analysis

The data for peat SOC was normally distributed and so comparisons between groups were analysed using ANOVA at $p<0.05$. The data for peat bulk density, depth and soil organic carbon density were not normally distributed, and so comparisons between the different peatland types were analysed using the non-parametric, Kruskal-Wallis test at $p<0.05$. ANCOVA was used to test for significant interaction between depth and peatland type when predicting total soil organic carbon density. All statistical analysis was carried out using SPSS (SPSS Inc., SPSS Statistics, Student Version, Release 17.0, 2008).

5.4. Results

5.4.1. Bulk density

For the eleven raised bog sites (RB1–RB11) the depth averaged bulk density ranged from 0.101 to 0.198 with a mean of 0.133 and standard deviation of 0.03 g cm^{-3} (Table 5.2). For the six high level blanket bog sites (HLB1-HLB6) the depth averaged bulk density ranged from 0.07 to 0.183 with a mean of 0.118 ± 0.04 g cm^{-3}. For the seven low level blanket bog sites (LLB1–LLB7) the depth averaged bulk density ranged from 0.088 to 0.177 with a mean of 0.125 ± 0.03 g cm^{-3}. The bulk density values reported here are within the range of values reported in the literature for non-forested peatlands. The bulk densities of the raised bogs are significantly higher than the bulk densities of the low level blanket bogs.

In general the bulk density for the upper 20 cm is higher than that at lower depths (Figure 5.2, 5.3 & 5.4). This is most likely due to shrinkage of the upper soil layer resulting from the drainage and water table lowering at the initial afforestation stage. The bulk density of each peat type shows little change with depth with some sites showing a decrease in bulk density with depth, with some sites, e.g. RB8 (Figure 5.2) showing an increase of bulk density with depth.

A number of bog sites show an increase in bulk density at the bottom of the peat profile. RB2 (Figure 5.2) has a bulk density value of 0.133 g cm^{-3} at 175 cm that increases to 0.183 g cm^{-3} at 225 cm. This increase is due to the presence of humic clays and clays at the bottom of the peat profile. The bulk density profiles of Tomlinson and Davidson (2000) and Weiss et al. (2002) show similar trends.

Of the 913 samples that were taken for bulk density across sites, RB1-RB5, HLB1-HLB4, and LLB1-LLB5, twelve contained pipes (connected natural conduits that transport water, sediment and solute through soil systems, Holden and Burt, 2002; Holden, 2006), three samples at LLB1, eight samples at HLB1 and one sample at RB2. The bulk density of samples with pipes was 0.07 g cm^{-3} which is significantly lower than the corresponding non-piped samples at 0.11 g cm^{-3} ($p<0.05$). Although the bulk density of the samples containing pipes is lower than adjacent samples, pipes are found in so few samples they were not significant to the overall mean bulk density of the peats.

Table 5.2. The mean ± standard deviation (SD), minimum and maximum for peat depth (cm), SOC (%) and bulk density (g cm^{-3}) for each site.

	Depth (cm)			SOC (%)			Bulk Density (g cm^{-3})		
Site	Mean ± SD	Min	Max	Mean ± SD	Min	Max	Mean ± SD	Min	Max
RB1	374 ± 20	335	409	53.6 ± 2	50.4	56.7	0.104 ± 0.01	0.095	0.13
RB2	180 ± 35	75	225	44.1 ± 11	23.9	50.8	0.128 ± 0.03	0.098	0.183
RB3	91 ± 24	42	134	43.5 ± 6	36.5	47.2	0.133 ± 0.02	0.116	0.149
RB4	87 ± 30	33	150	50.5 ± 1	49.1	51.2	0.127 ± 0.02	0.115	0.144
RB5	48 ± 15	30	84	34.6 ± 10	27.3	41.9	0.198 ± 0.001	0.199	0.198
RB6	345 ± 82	300	500	51.6 ± 2	48.0	55.3	0.101 ± 0.01	0.086	0.125
RB7	211 ± 47	115	282	43.8 ± 1	42.2	45.3	0.152 ± 0.04	0.107	0.215
RB8	129 ± 10	105	150	47.4 ± 2	45.0	49.4	0.158 ± 0.02	0.133	0.177
RB9	243 ± 12	226	267	48.5 ± 2	47.4	50.2	0.103 ± 0.02	0.086	0.148
RB10	217 ± 42	131	272	45.0 ± 3	39.6	47.7	0.148 ± 0.03	0.121	0.195
RB11	183 ± 9	164	198	51.1 ± 1	49.9	52.0	0.111 ± 0.02	0.098	0.133
RB	192 ± 100	30	500	46.7 ± 5	23.9	56.7	0.133 ± 0.03	0.086	0.215
HLB1	299 ± 65	105	419	49.8 ± 4	44.1	54.6	0.100 ± 0.01	0.087	0.114
HLB2	73 ± 20	30	100	50.7 ± 3	48.5	52.8	0.129 ± 0.01	0.12	0.139
HLB3	48 ± 14	32	89	48.9 ± 1	48.0	49.8	0.070 ± 0.01	0.065	0.075
HLB4	38 ± 6	30	50	38.3 ± 3	36.3	44.4	0.183 ± 0.03	0.123	0.208
HLB5	311 ± 42	250	375	53.8 ± 2	51.0	55.7	0.085 ± 0.01	0.071	0.109
HLB6	104 ± 21	60	130	51.4 ± 3	48.6	53.9	0.140 ± 0.01	0.126	0.151
HLB	145 ± 130	30	419	48.8 ± 5	36.3	55.7	0.118 ± 0.04	0.065	0.208
LLB1	336 ± 147	108	600	48.4 ± 3	44.3	52.3	0.088 ± 0.01	0.069	0.11
LLB2	113 ± 71	30	225	47.5 ± 2	44.4	49.8	0.122 ± 0.01	0.114	0.126
LLB3	90 ± 40	33	188	50.3 ± 2	48.5	52.2	0.096 ± 0.01	0.091	0.109
LLB4	98 ± 22	63	150	48.4 ± 8	39.6	54.1	0.131 ± 0.03	0.111	0.169
LLB5	36.7 ± 4	31	48	47.9 ± 2	45.4	49.8	0.154 ± 0.03	0.132	0.202
LLB6	60 ± 16	35	85	46.1 ± 4	43.5	48.8	0.177 ± 0.02	0.164	0.19
LLB7	158 ± 11	143	178	53.3 ± 2	50.8	55.4	0.104 ± 0.004	0.098	0.107
LLB	127 ± 100	30	600	48.8 ± 2	39.6	55.4	0.125 ± 0.03	0.069	0.202

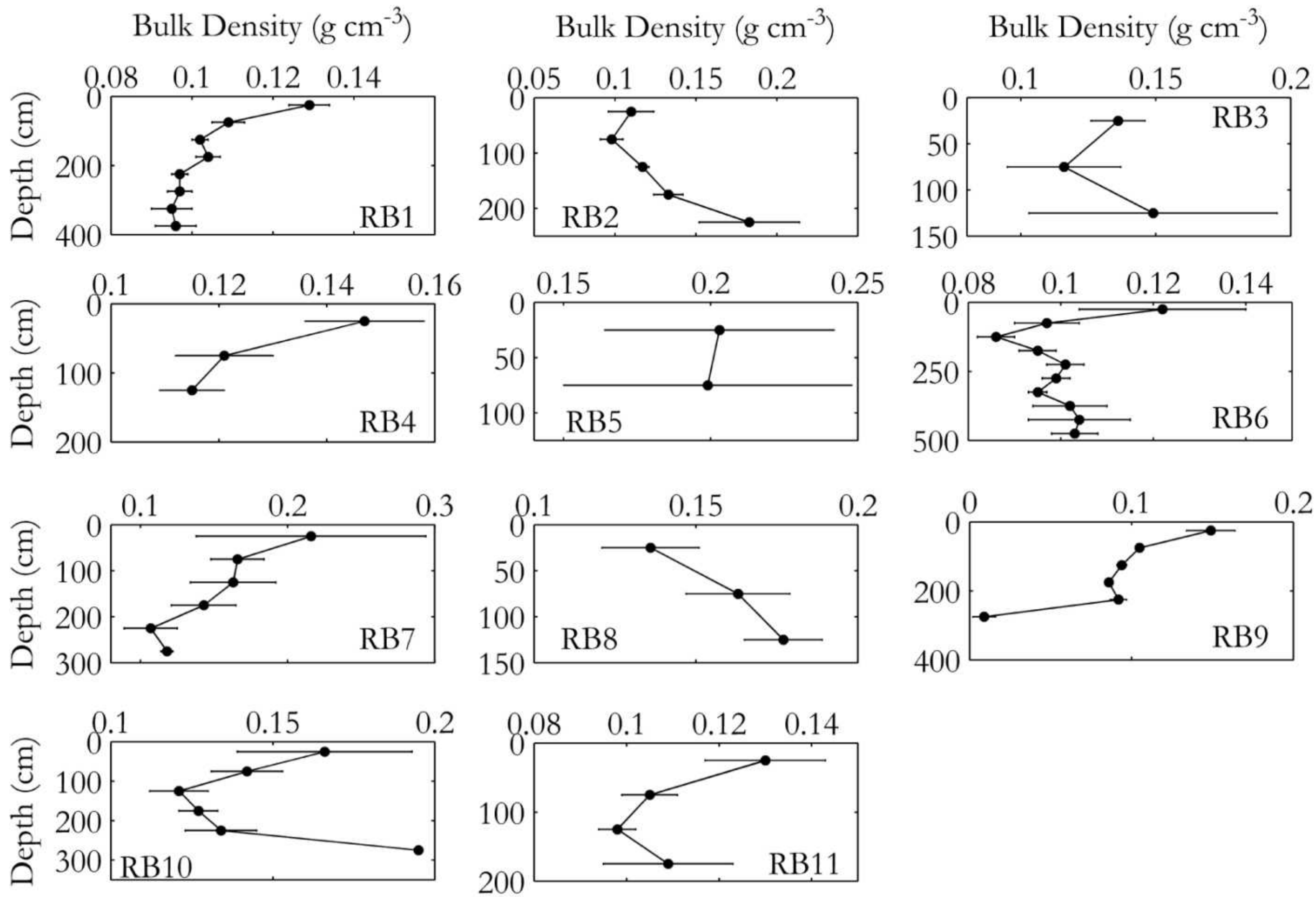

Figure 5.2. Bulk density (g cm-3) versus depth for the eleven afforested raised bog sites, RB1; RB2; RB3; RB4; RB5; RB6: RB7; RB8; RB9; RB10; RB11. Error bars indicate standard deviation.

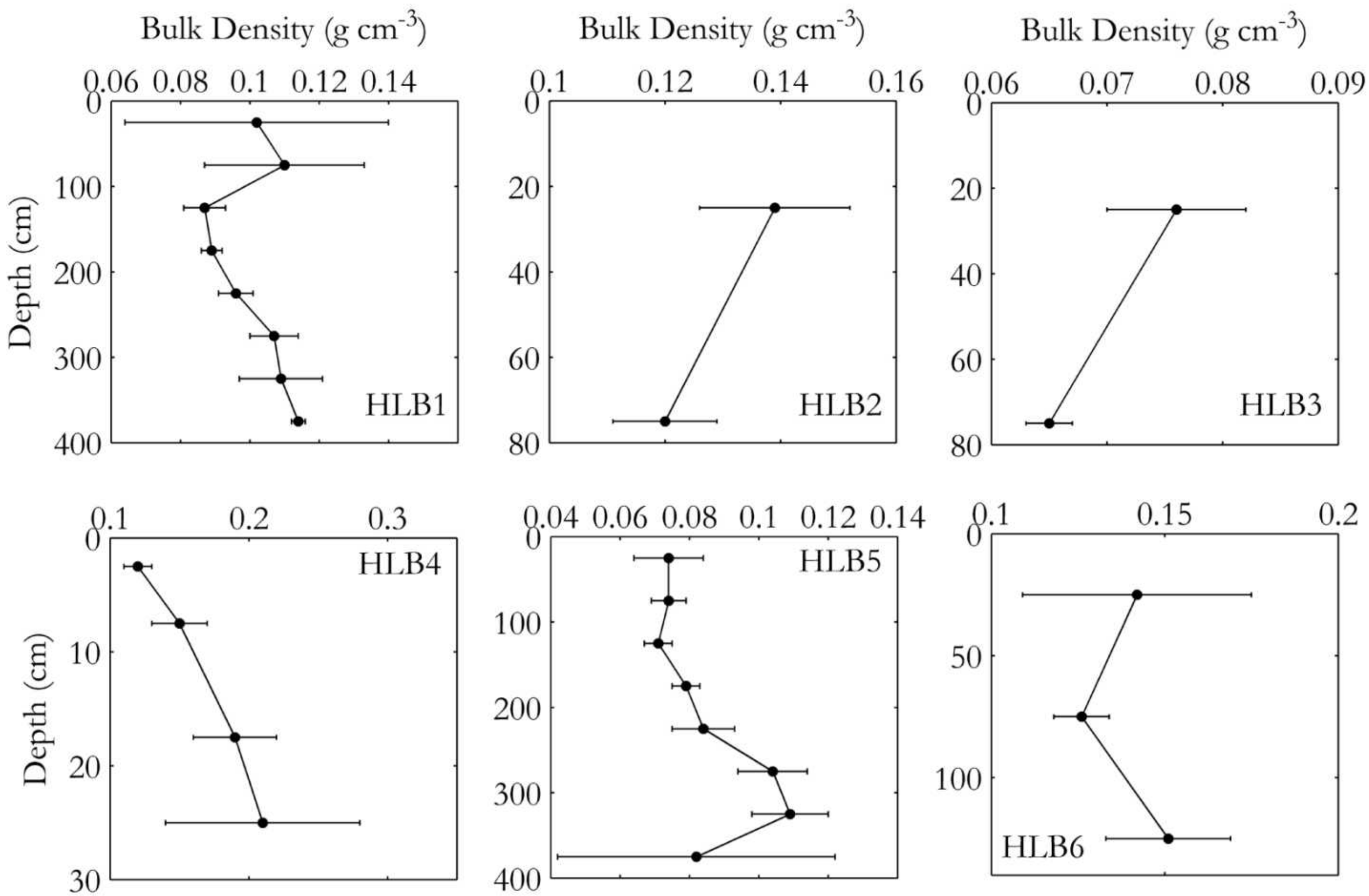

Figure 5.3. Bulk density (g cm-3) versus depth for the six afforested high level blanket bog sites, HLB1; HLB2; HLB3; HLB4; HLB5; HLB6. Error bars indicate standard deviation.

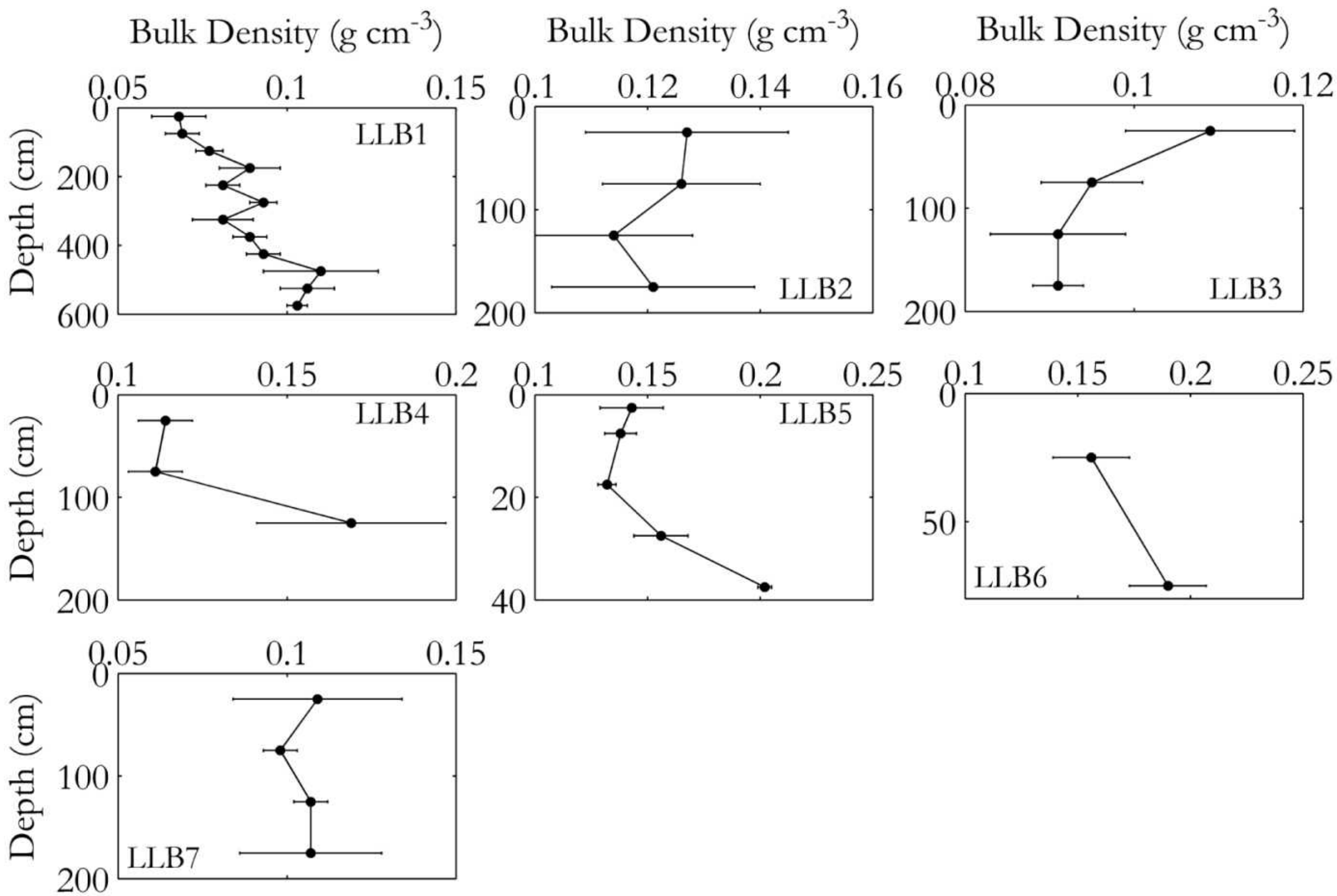

Figure 5.4. Bulk density (g cm^{-3}) versus depth for the seven afforested low level blanket bog sites LLB1; LLB2; LLB3; LLB4; LLB5; LLB6; LLB7. Error bars indicate standard deviation.

5.4.2. Soil organic carbon concentration (%)

The mean of the soil organic carbon (SOC) concentrations of the peat was 46.7% ± 5, 48.8% ± 5 and 48.8% ± 2 for raised, high level and low level blanket bogs, respectively. There are no significant differences in SOC concentrations between peat types. The presence of humic clays and clays at the lowest depths affects the SOC values, typically with an SOC reduction at the bottom of the peat profile. This is shown in LLB4 (Figure 5.5), where the SOC drops from 54.1% at 75 cm to 39.6% at 125 cm.

A number of the sites, especially within the high level and low level blanket bogs show the highest SOC concentrations at mid depth, with lower values near the surface and also at the bottom of the profile (Figure 5.5). The reduced SOC at the bottom of the peat profile is most likely due to mixing with the underlying mineral material.

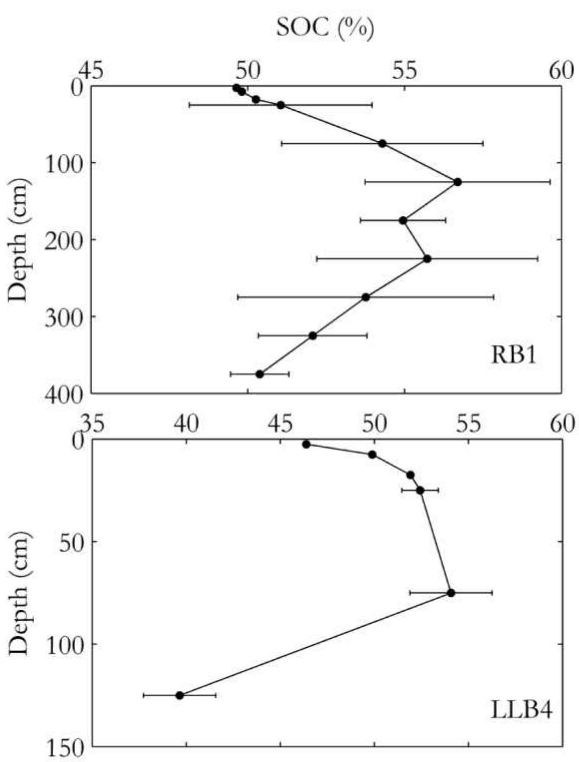

Figure 5.5. Soil Organic Carbon (SOC, %) versus depth for sites RB1; HLB1; LLB4. Error bars indicate standard deviation.

5.4.3. Peat depth

The mean depth varies widely, both within a site and between sites for all peat types (Table 5.2). For the eleven raised bog sites (RB1-RB11) the mean depth ranged from 48 to 374 cm with an overall mean depth of 192 ± 100 cm. For the six high level blanket bog sites (HLB1-HLB6) the mean depth ranged from 38 to 311 cm with an overall mean depth of 145 ± 130 cm. For the seven low level blanket bog sites (LLB1-LLB7) the mean depth ranges from 37 to 336 cm with an overall mean depth of 127 ± 100 cm. The depth of raised peat is significantly greater than either the high level or low level blanket bogs (p<0.05).

5.4.4. Soil organic carbon density

The soil organic carbon densities (SOCD) range from 180 Mg C ha^{-1} at site HLB3 (peat depth 48 cm) to 2090 Mg C ha^{-1} at site RB1 (peat depth 374 cm) (Table 5.3). Due to their greater depth, raised bogs have the largest mean soil organic carbon density of 1160 ± 520 Mg C ha^{-1}. The low level blanket bogs mean SOCD is 705 ± 420 Mg C ha^{-1} while the high level blanket bogs mean SOCD is 775 ± 590 Mg C ha^{-1}. The standard deviations of the SOCD means are very large due to the large variation of depth for each peat type. The SOCDs of the raised peat are significantly larger than those of both the high level and low level blanket bogs (p<0.05). The depth and peat type have a significant effect on the SOCD of each site (p<0.05) and are included in the following linear regression models to predict SOCD based on peat depth and peat type. The high level and low level blanket bogs were analysed together due to similar values of soil organic carbon density.

$$C_{RBC} = 4.878h + 228.604 \qquad R^2 = 0.95 \tag{5.4}$$

$$C_{BLC} = 4.414h + 138.399 \qquad R^2 = 0.96 \tag{5.5}$$

where: C_{RBC} = raised bog soil organic carbon density (Mg C ha^{-1}); C_{BLC} = blanket bog soil organic carbon density (Mg C ha^{-1}); h = depth of peat $\geq$ 30 cm;

There was no significant interaction between peat type and depth on the soil organic carbon density using ANCOVA, (p<0.05).

Table 5.3. The mean ± standard deviation soil organic carbon density (Mg C ha^{-1}) of all sites down the peat profile.

Site	Depth (cm)							Total
	0-50	50-100	100-200	200-300	300-400	400-500	500-600	
RB1	326 ± 15	295 ± 33	575 ± 32	531 ± 30	362 ± 94	-	-	2090 ± 140
RB2	272 ± 34	246 ± 33	559 ± 84	26 ± 50	-	-	-	1100 ± 96
RB3	314 ± 28	181 ± 150	54 ± 64	-	-	-	-	548 ± 200
RB4	358 ± 28	275 ± 110	98 ± 110	-	-	-	-	731 ± 190
RB5	368 ± 60	20 ± 44	-	-	-	-	-	388 ± 74
RB6	304 ± 44	253 ± 38	476 ± 34	547 ± 34	124 ± 160	103 ± 150	-	1800 ± 430
RB7	411 ± 93	376 ± 80	568 ± 140	128 ± 140	-	-	-	1480 ± 330
RB8	331 ± 39	387 ± 75	235 ± 110	-	-	-	-	952 ± 170
RB9	364 ± 44	262 ± 14	446 ± 20	193 ± 41	-	-	-	1270 ± 76
RB10	392 ± 67	339 ± 52	523 ± 110	144 ± 130	-	-	-	1400 ± 220
RB11	323 ± 30	274 ± 30	436 ± 69	-	-	-	-	1030 ± 100
HLB1	257 ± 85	272 ± 100	476 ± 41	480 ± 73	134 ± 120	-	-	1620 ± 250
HLB2	355 ± 47	155 ± 85	-	-	-	-	-	510 ± 100
HLB3	162 ± 22	18 ± 32	-	-	-	-	-	180 ± 49
HLB4	253 ± 73	-	-	-	-	-	-	253 ± 73
HLB5	189 ± 28	196 ± 27	400 ± 34	456 ± 130	119 ± 140	-	-	1360 ± 200
HLB6	383 ± 88	272 ± 72	72 ± 82	-	-	-	-	727 ± 170
LLB1	162 ± 19	175 ± 26	370 ± 60	329 ± 140	251 ± 140	195 ± 160	74 ± 120	1550 ± 640
LLB2	249 ± 54	200 ± 150	233 ± 190	-	-	-	-	682 ± 380
LLB3	263 ± 21	137 ± 100	61 ± 89	-	-	-	-	460 ± 170
LLB4	294 ± 21	265 ± 78	73 ± 94	-	-	-	-	632 ± 130
LLB5	255 ± 7	-	-	-	-	-	-	255 ± 7
LLB6	362 ± 27	106 ± 120	-	-	-	-	-	468 ± 130
LLB7	283 ± 56	273 ± 26	325 ± 54	-	-	-	-	881 ± 92

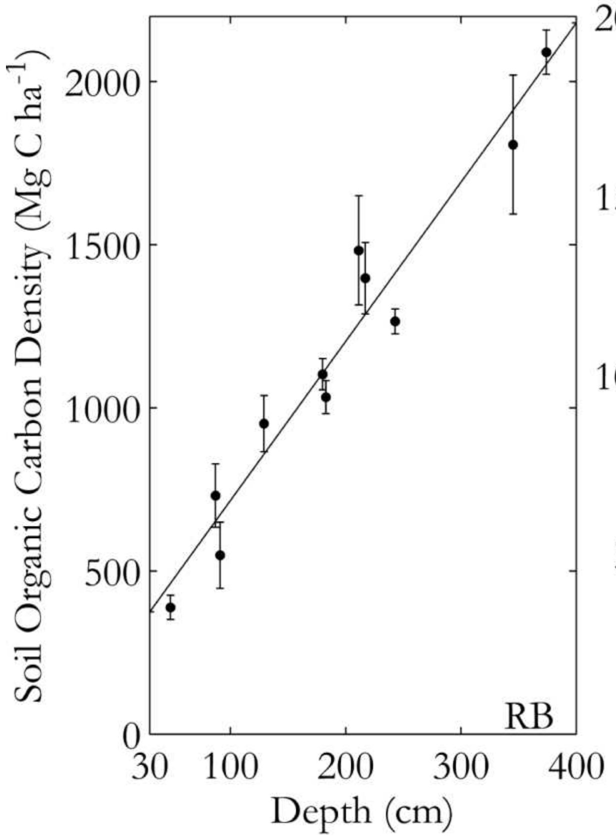

Figure 5.6. The soil organic carbon density of each peat type, RB, raised bog; HLB, high level blanket bog and LLB, low level blanket bog, versus depth. The regression equations, 5.4 and 5.5 are presented. $C_{RBC} = 4.878h + 228.604$ $R^2 = 0.95$. $C_{BLC} = 4.414h + 138.399$ $R^2 = 0.96$. Where C_{RBC} = raised bog soil organic carbon density (Mg C ha^{-1}); C_{BLC} = blanket bog soil organic carbon density (Mg C ha^{-1}); h = depth of peat $\geq$ 30 cm. Error bars indicate standard deviation.

5.5. Discussion

5.5.1. Peat bulk density and SOC concentration

The values for bulk density are within the range reported in the literature for forested and non-forested peatlands. There seems to be little difference in bulk density between non-forested and afforested peatlands below the 20 cm depth. For the near-surface depth (up to 20 cm) the afforested peatlands seem to be denser, most likely due to compaction and water table lowering caused by drainage planting and increased evapotranspiration from the trees (Minkkinen et al., 2008).

Tomlinson and Davidson (2000) observed a mean bulk density of 0.069 g cm^{-3} for non-forested raised bogs in Northern Ireland which is much lower than our mean value of 0.136 g cm^{-3}. Chapman et al. (2009) used bulk density values of raised peat at 0.112 g cm^{-3} lower than the values reported here. Chapman et al. (2009) found bulk density values of blanket peat in Scotland of 0.129 g cm^{-3} which are similar to our value of 0.121 g cm^{-3}. Burke (1978) found a mean bulk density of 0.097 g cm^{-3} for a forested blanket bog site in Ireland to 90 cm depth, which is less than our value of 0.121 g cm^{-3}. However, Shotbolt et al. (1998) found a mean bulk density value of 0.13 g cm^{-3} for a forested low level blanket bog site in Scotland, a value greater than our value of 0.125 g cm^{-3}. In contrast, Shotbolt et al. (1998) found a lower mean of 0.11 g cm^{-3} for an adjacent non-forested low level blanket bog site that is similar to the mean values of this study. The bulk densities of the raised bog sites are statistically greater than that of the low level blanket bogs, for the entire profile. It is unclear why this is the case, but is most likely due to the different processes and plant species which form each peat type.

It is widely noted in the literature that the bulk density of peat increases with depth (possibly attributed to the effect of greater humification over time) (Clymo, 1978; Howard et al., 1994; Milne and Brown, 1997; Cruickshank et al., 1998). However, this was not found in our study as there was very little change in bulk density, with variation between sites. Tomlinson and Davidson (2000) found no increase in non-forested raised bogs with depth in Northern Ireland. Weiss et al. (2002) in a peat bog in south east Asia, found an initial increase in bulk density in the top 150 cm of peat reflecting the transformation of living matter to poorly decomposed peat, and then from 200 cm onwards the bulk density decreases until it reaches the peat/sediment interface at 840 cm. Lewis et al. (2011) also found no change with depth in a high level non-forested blanket bog in Glencar, Ireland. The results of Lewis et al. (2011) are interesting in that at the centre of the bog (where the water table never fell below 15 cm) the bulk density of 0.055 g cm^{-3} was about half of that of 0.11 g cm^{-3} at the peat margins (a narrow 20m wide strip near a stream, where the water table was generally below 30 cm) highlighting the spatial variation of bulk density within the peatland. The increase in bulk density with depth due to compression of the physical structure that is often assumed in peatland carbon stock modelling may not be accurate, and may be leading to an overestimate of peatland carbon stocks, especially in deep peatlands.

The values of SOC reported here are lower than those reported in the literature. Tomlinson and Davidson (2000) found an SOC concentration of 51.1% for non-forested raised bogs in Northern Ireland. Lewis et al. (2011) found a mean SOC of 52.9% for a non-forested, pristine low level blanket bog. Our findings suggest that there may have been some minor losses in SOC (~3%) in the afforested peatlands, possibly due to loss of CO_2 to the atmosphere as a result of increased aeration due to lowering of the water table in the afforestation process.

The impact of afforestation on the physical properties of peat may be better evaluated by assessing the decomposition of the peat samples as well as the bulk density. Kechavarzi et al. (2010) shows that the bulk density of peat increases with increasing decomposition. Analyses of peat decomposition using the von Post scale (von Post, 1924) may explain some of the variation in bulk density with depth.

5.5.2. Peat depth and soil organic carbon density

The measurement of peat depth varies widely between nations with some nations i.e. Finland, having extensively measured peat depth, but globally and within Ireland there has been very little measurement (Lindsay, 2010). Tomlinson (2005), in his estimate of soil carbon stocks in Ireland, used mean depths (taken from a number of small databases from the literature and businesses) of 60 cm and 100 cm for high and low level, man modified (man modified encompasses a number of land use types, including forested peatlands) blanket bogs which are lower than our results of 145 cm and 127 cm respectively. Tomlinson (2005) used a depth of 150 cm for western Ireland raised bogs and 250 cm for midland Ireland raised bogs in contrast to our results for western raised bogs of 172 cm for sites RB1, RB2, RB4 and RB5 and 183 cm for our midland raised bogs, of sites RB3, RB9, RB10 and RB11. The values presented here represent the first comprehensive field sampling analysis of the soil organic carbon density of Irish peatland soils. Tomlinson (2005) used soil organic carbon densities based on a very small sample size, using a mean of 1290 Mg C ha^{-1} for man-modified raised bogs, slightly bigger than the 1160 Mg C ha^{-1}, reported here. However, our estimates for the low level blanket and high level blanket bogs, 705 and 775 Mg C ha^{-1}, were higher than those of Tomlinson (2005) of 585 Mg C ha^{-1} and 270 Mg C ha^{-1} respectively.

High level blanket bogs are the largest forested peat type, representing 49.9% of the total forested area. The published soil organic carbon density estimates (Tomlinson, 2005) for forested high level blanket peat, at 270 Mg C ha^{-1} is much lower than our value of 772.2 Mg C ha^{-1}. Tomlinson (2005) estimated that the effect of human activities on peatlands would see a depth decrease of 66% for raised and low level blanket bogs and 50% for the high level blanket bogs. The differences between our values and the estimates of Tomlinson (2005) suggest that to improve future peatland carbon stock estimates, afforested peat carbon stocks should be calculated separately from non-forested peat. By incorporating the soil organic carbon density estimates of this study with the forest cover area from NFI (2007a) and using updated peatland maps from Connolly et al. (2007), the estimates of peatland carbon stocks will be much improved. Peat depth is a key property of the afforested peat soil organic carbon densities for each peat type (Figure 5.6). Depth is the simplest and quickest property to measure and the estimates based on equations 5.3 and 5.4, will allow for a much quicker assessment of peat soil organic carbon densities. However, much fieldwork is required to get a more detailed picture of the spatial variation of peat depth as we know from Lewis et al. (2011) that there is significant spatial variability within the km scale and less within the same peatland.

Equations 5.4 and 5.5 are unlikely to be appropriate for non-forested peatlands and boreal forested peatlands, where most of the global forested peatland is situated due to lower values of bulk densities found at these sites (Minkkinen and Laine, 1998; Minkkinen et al., 1999; Tomlinson and Davidson, 2000; Anderson, 2002; Lewis et al., 2011). Equations 5.4 and 5.5 could be used as an estimate of soil organic carbon density for other afforested peatlands in Ireland and Britain due to the similar peat types and climate, however, care must be taken in using them as these sites only measured a small plot of each forest stand and do not measure the variation within the larger peatland. The methodology may be used to create similar equations for other forested and non-forested peatland in other nations.

The depth and soil organic carbon density of the raised bogs are significantly larger than both the high and low level blanket bog sites as expected. For future estimates of peat soil organic carbon density and accounting purposes, raised and blanket bogs should be sampled and reported separately. The linear equations and soil organic carbon densities of the high and low level blanket bogs are similar with 775 Mg C ha^{-1} and 705 Mg C ha^{-1}, respectively. The high level and low level blanket bogs can be considered as the same peat type due to similar peat soil organic carbon densities, and analysed simply as blanket bogs in future work.

5.6. Conclusions

We present the depths, bulk densities, soil organic carbon (SOC) concentrations and soil organic carbon densities of 24 Irish afforested peatlands. To date few publications exist on the physical properties of forested peat which limits the estimates of peatland carbon stocks (Tomlinson, 2005; Limpens et al., 2008; Chapman et al., 2009). The data presented in this study can be used to improve estimates of afforested peatland carbon stocks.

The view of peat bulk density increasing with depth is not supported by our work as each peat type showed little change down the profile, with individual sites showing different trends. There is an increased bulk density and lower SOC in the top 20 cm of the peat, possibly due to the lowering of the water table and the subsequent increased aeration of the peat due to drainage preparations and afforestation.

The depths of peat and thus the carbon stocks of forested peatland may have been underestimated for Ireland in Tomlinson (2005), especially for blanket bogs where we found depths of 145 and 127 cm for high level and low level compared to the 60 and 100 cm. High level blanket bog represents 49.9% of the total Irish forested peatland area; therefore any underestimate of the carbon stocks represents an underestimate of the carbon stocks of the entire national forest area. The underestimate of the forested peatland carbon stocks also shows that in future estimates of soil carbon stocks, forested peatlands should be analysed separately from other non-forested peatlands. The similar values of soil organic carbon density show that afforested high level and low level blanket bogs can be grouped together and analysed simply as blanket bogs. The depth and soil organic carbon densities of raised bog sites are significantly larger than those of the blanket bogs, and so further analysis of peat soil organic carbon density should focus on both the raised and blanket bogs.

The linear regression equations of soil organic carbon density with depth and peat type are a simple first estimate of soil organic carbon density of afforested peatlands for Ireland and Britain. The linear regression equations require knowledge only of the peat type and the peat depth to estimate the soil organic carbon density. The methodology presented here can be used to create similar linear regression equations for non-forested peatlands. It is important that when modeling the carbon stocks of peatlands, there should be a greater analysis of the physical properties of all peat types, especially depth, to improve the reliability of the estimates.

5.7. Acknowledgements

We thank Hadrien Robin, Gaelle Treut, Sylvia Dolan and Rachel Wisdom for their help with the field sampling and the laboratory analysis. This work was supported by two projects "Soil Carbon Stock Changes and Greenhouse Gas Flux in Irish Forests (ForestC)" and "Carbon sequestration by Irish forest ecosystems (CARBiFOR II) contributed to this study. Both are funded by the Irish Council for Forest Research and Development (COFORD) as part of the Department of Agriculture Fisheries and Food (DAAF).

6. Changes in ecosystem carbon stocks in a grassland-ash (Fraxinus excelsior L.) afforestation chronosequence in Ireland

Michael L. Wellock[1*], Christina M. LaPerle[1], Matthias Peichl[1], & Gerard Kiely[1]

[1] *Centre for Hydrology, Micrometeorology and Climate Change, Department of Civil and Environmental Engineering, University College Cork, Cork, Ireland.*

Submitted to European Journal of Forest Research February 3rd 2011 and is currently under review.

Keywords: ash (*Fraxinus excelsior* L.), chronosequence, soil, biomass, carbon, ecosystem.

6.1. Abstract

Government policy in Ireland is to increase the national forest cover from the current 10% to 17% of the total land area by 2030. This represents a major land use change that is expected to impact on the national carbon (C) stocks. While the C stocks of ecosystem biomass and soils of Irish grasslands and coniferous forests have been quantified, little work has been done to assess the impact of broadleaf afforestation on C stocks. In this study we sampled a chronosequence of ash (*Fraxinus excelsior L.*) forests aged 12, 20, 27, 40 and 47 years on brown earth soils. A grassland site, representative of the pre-afforestation land use, was sampled as a control. Our results show that there was a significant decline ($p<0.05$) in the soil organic carbon density ($SOCD_i$, 0-30 cm) following afforestation from the grassland (90.2 Mg C ha^{-1}) to the 27 year old forest (65.9 Mg C ha^{-1}). Subsequently, the forest soils switched from being a C source to a C sink, and began to sequester C to 70.6 Mg C ha^{-1} at the 47 year old forest. We found the carbon density of the above- and belowground biomass increased with age of the forest stands, by 1.83 Mg C ha^{-1} yr^{-1} and offset the amount of C lost from the soil. The increased biomass carbon density led to an increase in the ecosystem carbon density, from 92.6 Mg C ha^{-1} at the grassland site to 161.9 Mg C ha^{-1} at the 47 year old forest. This creates the potential to sequester significant amounts of C with ash afforestation in the coming years. The effects of harvesting and reforestation may further modify the development of ecosystem carbon density over an entire ash rotation.

6.2. Introduction

At the beginning of the twentieth century, forests accounted for only 1% of the total Irish land cover (Pilcher and Mac an tSaoir, 1995). However, due to the facilitating efforts of successive governments there has been rapid afforestation since the 1960s resulting in 10% forest land cover as of 2007 (NFI, 2007a). A large proportion of this afforestation took place after the mid-1980s and was fueled by government grant incentive schemes targeted at private landowners. Consequently, 63% of forests are less than 20 years old (NFI, 2007a). This specific land use change provides an opportunity for Ireland to meet in part its international obligations set forth by the United Nations Framework Convention on Climate Change (UNFCCC, 1992), as article 3.3 of the Kyoto Protocol allows changes in carbon (C) stocks due to afforestation, reforestation, and deforestation since 1990 to be used to offset inventory emissions (Kyoto Protocol, 1997). To assess if Ireland can benefit from Article 3.3, the impact of afforestation on the C stocks of Ireland must be determined.

Before the 1990s, broadleaf afforestation had been low, at approximately 3-4% of total annual afforestation. Conifers make up the majority of afforestation, especially Sitka spruce (*Picea sitchensis* [Bong] *Carr*.), which are preferred due to their greater productivity. Broadleaf planting increased to 20% of total annual afforestation in 1995, but steadily decreased to 13% in 2000 (Renou and Farrell, 2005). It is current policy that broadleaves should make up 30% of new plantings.

Ash (*Fraxinus excelsior* L.) is one of the most popular species of indigenous broadleaf in Ireland, as it has a short economic rotation length for a broadleaf of 60 to 80 years. Ash is also used in the indigenous industry of hurley making as used in the Irish sport of hurling (Joyce et al., 1998; Horgan et al., 2004). There are currently 19,200 ha of ash forests in Ireland (NFI, 2007a), and with the government's policy to increase forest cover from its current 10% (NFI, 2007a) to 17% by 2030 (Department of Agriculture, 1996), and that 30% of new forests should be broadleaves, the amount of land expected to be forested with ash will greatly increase.

It is important to assess the impact of afforestation on soil organic carbon (SOC) stocks as they provide a long term store of C which is longer than that of the C stored in the forest biomass (Vesterdal et al., 2002; Black et al., 2009). Irish forest soils are estimated to contain between 6 and 11% of the national total soil C stock (Tomlinson, 2005; Eaton et al., 2008; Xu et al., 2011).

Soil C stocks are controlled by the balance between the inputs of organic material from the biota, and losses, primarily from microbial respiration (Zerva and Mencuccini, 2005; Peng et al., 2008). The change in soil C stocks depends on myriad factors including: tree species, soil type, rotation length, age, site management, topography and climate (Dixon et al., 1994; Thuille et al., 2000; Jandl et al., 2007). Some studies have found no change in SOC stocks after afforestation (Davis et al., 2003; DeGryze et al., 2004; Smal and Olszewska, 2008; Peri et al., 2010). Others have found an increase in SOC stocks (Post and Kwon, 2000; Guo and Gifford, 2002; Hooker and Compton, 2003; Morris et al., 2007; Black et al., 2009; Mao et al., 2010) while some have found a decrease in SOC stocks (Parfitt et al., 1997; Alberti et al., 2008). However, many studies have shown the same trend, a loss of SOC stocks in the initial years after afforestation, but thereafter increasing with age to match pre-afforestation levels and even to surpass them (Romanyà et al., 2000; Parker et al., 2001; Turner et al., 2005; Ritter, 2007).

While the effects of afforestation on soil C stocks vary between studies, there is a consensus that there is a large increase in the C stored in the above- and belowground biomass following afforestation. The amount sequestered into the forest biomass increased on average by 1.18 -1.53 Mg C ha^{-1} yr^{-1}, whereas the C sequestered into the soil is much less and varies from a C sink to a source, -0.69 to +0.15 Mg C ha^{-1} yr^{-1} (Hooker and Compton, 2003; Alberti et al., 2008). Other studies have found large increases in the forest biomass C following afforestation (Vesterdal et al., 2002; Tremblay et al., 2006; Taylor et al., 2007; Black et al., 2009).

This study determined the ecosystem carbon density in five afforested ash chronosequence stands (12, 20, 27, 40, and 47 years) and adjacent grassland to investigate the development of ecosystem C storage following grassland afforestation. The aims of the study were: 1) to estimate the changes in bulk density (g cm^{-3}), SOC concentration (%) and soil organic carbon density (Mg C ha^{-1}) of the soil; and 2) to estimate the successional changes of C stored in the tree biomass and forest floor.

6.3. Materials and methods

6.3.1. Site description

The study sites were selected from private and state owned forests located in central Ireland. Five afforested ash stands were selected for sampling; aged 12, 20 27, 40 and 47, and referred to as F12, F20, F27, F40 and F47, respectively. Prior to afforestation, each forest site was grassland. A grassland site adjacent to site F12 was sampled to provide a comparison to all sites and will be referred to as G0. The sites, G0 and F12 were located on private land within County Offaly, while sites F20, F27, F40 and F47 were located within the state owned Coillte forest of Castlemorris in County Kilkenny (Figure 6.1, Table 6.1). The forest sites consist of similar aged stands of ash, ranging in elevations between 73 to 106 m above sea level and with flat topographies. Each site has a brown earth soil: a well-drained, mature mineral soil possessing a uniform profile (Gardiner and Radford, 1980). The chronosequence technique used in this study allows for the integration of independent forest stands with different ages, into one unit, substituting space for time. The stands are as close to identical in all aspects other than stand age so as to reduce variability (Taylor et al., 2007; Hedde et al., 2008; Black et al., 2009; Tang et al., 2009).

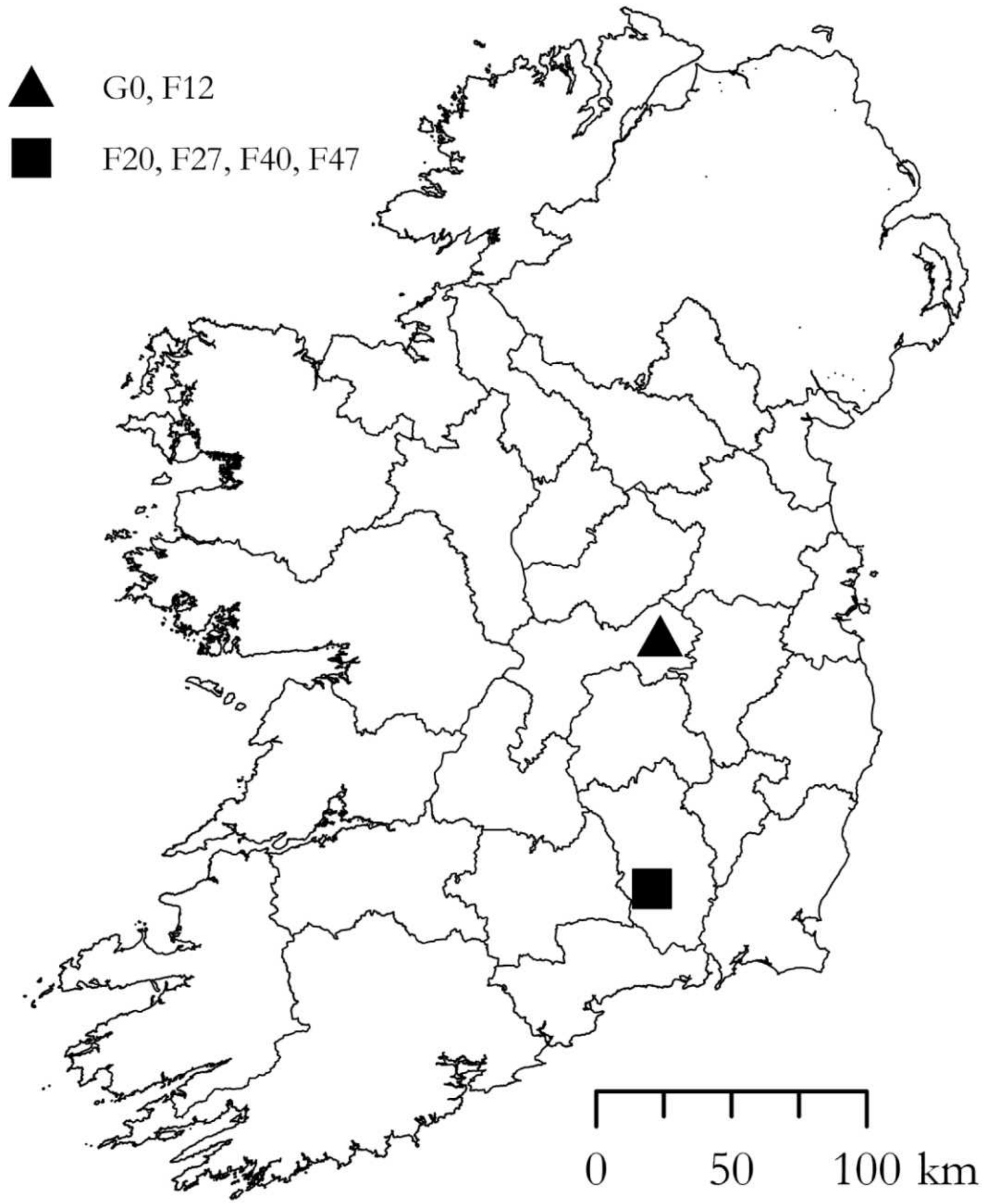

Figure 6.1. Locations of all 6 sites within Ireland.

Table 6.1. Characteristics of each site.

Forest Site	Stand Age (years)	Elevation (m)	Georeference Position	Stocking Density (Stem ha^{-1})	DBH (cm)
G0	-	73	53° 17' N, 7° 12' S	-	-
F12	12	73	53° 18' N, 7° 12' S	3433 ± 551	8.7 ± 0.1
F20	20	87	52° 27' N, 7° 16' S	1400 ± 131	14.5 ± 0.4
F27	27	75	52° 27' N, 7° 16' S	787 ± 70	17.6 ± 0.7
F40	40	106	52° 27' N, 7° 16' S	727 ± 214	23.8 ± 3.0
F47	47	102	52° 27' N, 7° 16' S	453 ± 153	26.6 ± 3.5

6.3.2. Soil and forest floor sampling methodology

Sites G0 and F12 were sampled in February 2010, while sites, F20, F27, F40 and F47 were sampled in June and July 2010. At each site a 20 m by 20 m square plot was set out, and then partitioned into four 10 m by 10 m quadrants. The soil at each site was sampled to 30 cm depth. Within each quadrant, 4 points were randomly selected and sampled for bulk density (g cm^{-3}) at the depths of 0-5, 5-10, 15-20 and 25-30 cm. Horizontal spacing of 5 cm were used between samples below 10 cm to avoid the effects of compaction from previous samples. Bulk density samples were taken using stainless steel bulk density rings (Eijkenkamp Agrisearch Equipment BV, Netherlands) of 8 cm diameter by 5 cm height.

Adjacent to the point where the bulk density samples were taken, 8 holes were sampled to depths of 0-10, 10-20 and 20-30 cm for soil organic carbon using a 10 cm soil auger (Eijkenkamp Agrisearch Equipment BV, The Netherlands).

At three points within each quadrant, the forest floor was sampled from 0.1 m^2 square plots. Each forest floor and soil sample was placed into a labeled polythene bag in the field.

All soil samples were stored at 4 °C before being dried at 55 °C, until a constant dry weight was achieved. The soil samples taken with the soil auger for SOC were sieved to <2 mm (the 8 soil samples were bulked for each depth at each quadrant by equal volume) and then ground to a fine powder. All soil samples were tested for carbonates. Any samples that tested positive for carbonates were treated with sulphurous acid to remove the carbonates (Nelson and Sommers 1996) Sites G0 and F40 were found to have carbonates present. Samples were combusted in a C/N analyser (Elementar - Vario Max CN) to determine the SOC (%).

The bulk density was calculated using equations (6.1).

$$\rho_d = \frac{MA}{SV - CFV} \tag{6.1}$$

where ρ_d = bulk density (g cm^{-3}); MA = mass of dry sample <2 mm (g); SV = sampler volume (cm^3); CFV = >2 mm coarse fraction volume (cm^3)

The bulk density samples were converted to 10 cm depths to match the SOC data. The volume based method is often used in assessing changes in soil organic carbon density following land use or management changes by multiplying the SOC concentration (%) by bulk density (g cm^{-3}) to a fixed depth. The volume-based method assumes that bulk density is static through time (Markewitz et al., 2002; Sartori et al., 2007; Mao et al., 2010), however, bulk density varies spatially and temporally (Lee et al., 2009) and so changes in soil mass can mask changes in SOC and produce erroneous results. The equivalent soil mass method (ESM) was designed so as to remove this error in measurements of SOCD change and is defined as the reference soil mass per unit area chosen in a layer (Ellert et al., 2001). The soil mass was calculated using equation (6.2).

$$M_i = \rho_{di} \times h \times 10^4 \tag{6.2}$$

where M_i = dry soil mass (Mg ha^{-1}) at the i^{th} depth (i =1,2,3 corresponding to the 0-10, 10-20 and 20-30 depths); h = depth of soil layer (cm); 10^4 = unit conversion factor (m^2 ha^{-1}).

The soil organic carbon density of the soil to a fixed depth using the volume-based method was calculated using equation (6.3).

$$C_{i,vol} = conc_i \times M_i \tag{6.3}$$

where $C_{i,vol}$ = soil organic carbon density, volume based method (SOCDvol, Mg C ha^{-1}); $conc_i$ = SOC (%)

The equivalent soil mass method uses the soil mass of each soil layer sampled at the control site as the ESM for the layer, using equation (6.4), the equivalent carbon mass (ECM) can be calculated using equation (6.5) (Lee et al., 2009).

$$M_{i,add} = M_{i,equiv} - M_i \tag{6.4}$$

$$C_{i,esm} = C_{i,vol} - conc_{i+1} \times M_{i-1,add} + conc_{i-1} \times (M_{i,add} - M_{i-1,add}) \quad (6.5)$$

where $C_{i,esm}$ = the soil organic carbon density, equivalent soil mass method (SOCDesm, Mg C ha^{-1}); $M_{i,equiv}$ = non-forest soil mass (Mg ha^{-1}); $M_{i-1,add}$ and $M_{i,add}$ = the additional soil masses that are used to estimate the ESM (Mg ha^{-1}); $conc_{i-1}$ and $conc_{i-1}$ = SOC concentration (%) for the additional soil mass determined by the location of soil mass used for the corrections between the soil layers at each sampling time. For more information on the ESM method see Ellert and Bettany (1995), Ellert et al. (2001) and Lee et al. (2009).

The soil organic carbon density (0-30 cm) of the forest sites were calculated using equation (6.6) and the soil organic carbon density (0-30 cm) of the grassland site was calculated using equation (6.7)

$$C_{tot} = \sum C_{i,esm} \quad (6.6)$$

$$C_{tot} = \sum C_{i,vol} \quad (6.7)$$

where C_{tot} = soil organic carbon density (0-30 cm) using either the volume based method or equivalent soil mass method.

The soil texture was analysed at each depth for all sites from the bulked soil samples, using the hydrometer method, ASTM D422, 2002.

To summarise, at each of the 6 sites we obtained: (1) 32 samples for SOC at each depth of 0-10, 10-20 and 20-30 cm using the soil auger that were bulked into 4 samples for each depth for analysis; (2) 4 samples of bulk density of the depths, 0-5, 5-10, 15-20 and 25-30 cm using the stainless steel rings; and (3) 12 samples of the forest floor.

6.3.3. Above- and belowground biomass

The methodology used to measure the biomass of the sites was adapted from the Irish National Forest Inventory (NFI) Protocol (NFI, 2007b) and was non-destructive. Three random points were chosen within each forest site and a circular plot of 500 m^2 (25.24 m diameter) was set up with random points as plot centers. The diameter at breast height (cm) (DBH at 1.3 m) was recorded for each tree that had a DBH greater than 7 cm within each plot. Sites F40 and F47 were smaller than the other ash stands (<1500 m^2) and so the third plot was set to an area of

250m². The stem and branch biomass was calculated from an allometric equation (6.8) of ash trees from an English study (Bunce, 1968).

$$ln(S) = -2.4598 + 2.4882ln(d) \tag{6.8}$$

where S = stem and branch biomass (kg); d = DBH (cm)

The leaf biomass data were calculated from equation (6.9) of ash trees from an Italian study (Alberti et al., 2005).

$$L = 0.003d^{2.31} \tag{6.9}$$

where L = leaf biomass (kg); d = DBH (cm)

The root biomass was calculated from equation (6.10) (Cairns et al., 1997).

$$R = 0.26b \tag{6.10}$$

where R = root biomass (kg), b = aboveground biomass (kg).

All biomass and forest floor litter was multiplied by 0.5 to estimate the mass of C (IPCC, 1997). The forest floor carbon density was calculated using equation (6.11).

$$C_{ff} = 0.5FF \tag{6.11}$$

where C_{ff} = forest floor carbon density (FFCD, Mg C ha^{-1}), FF = forest floor mass (Mg ha^{-1}), 0.5 = conversion factor to convert into carbon density.

The carbon density of the above-ground biomass was calculated using equation (6.12).

$$C_a = (S \times ST + L \times ST) \times 0.5 \times 10^{-3} \tag{6.12}$$

where C_a = aboveground biomass carbon density (AGCD, Mg C ha^{-1}), ST = stocking density (stem ha^{-1}), 0.5 = conversion factor to convert into carbon density, 10^{-3} is a conversion factor.

The carbon density of the below-ground biomass was calculated using equation (6.13).

$$C_w = 0.26 C_a \tag{6.13}$$

where C_w = belowground biomass carbon density (BGCD, Mg C ha^{-1}).

The carbon density of the biomass was calculated using equation (6.14).

$$C_b = C_a + C_w \tag{6.14}$$

where C_b = biomass carbon density (BCD, Mg C ha^{-1}),

The carbon density of the ecosystem was calculated using equation (6.15).

$$C_e = C_b + C_{ff} + C_{tot} \tag{6.15}$$

where C_e = ecosystem carbon density (Mg C ha^{-1}).

6.3.4. Statistical analysis

The biomass carbon density, soil bulk density and SOC data, along with the 20-30 cm depth for soil organic carbon density were normally distributed and so were analysed by one way ANOVA to test for significant differences between all sites. The 0-10, 10-20 and 0-30 cm depths for soil organic carbon density and forest floor carbon density values were not normally distributed, and were analysed using the non-parametric test, Kruskal-Wallis. All statistical analysis was calculated at p<0.05 using SPSS (SPSS Inc., SPSS Statistics, Student Version, Release 17.0, 2008).

6.4. Results

6.4.1. Soil texture and soil organic carbon

The soil texture class of loam was the same at all sites (Table 6.2). There is no difference in the soil texture along the chronosequence, confirming the suitability of the soils for sampling.

For the 0–10 cm depth, the SOC (mean ± standard deviation) decreased from its highest value of 4.5% ± 0.8% at G0 with increasing stand age to a minimum of 3.4 ± 0.5% and 3.4 ± 0.2%, at F27 and F40, respectively (Table 6.3). At the later stand age, the SOC was significantly lower compared to G0 and F12 (p<0.05). The SOC was higher at F47 than it was at F27 and F40, but still lower than G0 and F12, although the SOC at F47 was statistically different only to the SOC at G0. The SOC of the 10-20 cm and 20-30 cm depths followed similar patterns as the 0-10 cm, although the magnitudes of SOC decreased with depth for sites. For example the SOC for F12 decreased from 4.3% at 0-10 cm to 2.5% at 10-20 cm and to 1.6% at 20-30 cm giving an overall depth average of 2.8% for the 0-30 cm depth.

Table 6.2. The soil texture as % sand, silt and clay (mean and standard deviation, SD) at each site; n = 3 for each site.

Site	Sand (%)	Silt (%)	Clay (%)	Soil Texture Class
G0	43.4 ± 3.6	34.5 ± 8.2	22.1 ± 4.8	Loam
F12	39.9 ± 4.1	35.1 ± 1.6	25.0 ± 5.0	Loam
F20	40.8 ± 1.7	33.9 ± 2.0	25.3 ± 0.7	Loam
F27	39.7 ± 1.9	37.6 ± 1.7	22.7 ± 3.1	Loam
F40	45.9 ± 2.6	33.8 ± 1.1	20.3 ± 3.1	Loam
F47	43.6 ± 3.0	36.1 ± 1.6	20.4 ± 2.6	Loam

Table 6.3. The mean, standard deviation (SD), minimum and maximum values of bulk density (BD, g cm^{-3}), SOC (%) and soil organic carbon density (equivalent soil mass method) (SOCDesm, Mg C ha^{-1}) and soil organic carbon density (volume-based method) (SOCDvol, Mg C ha^{-1}) for each soil depth of 0-10 cm, 10-20 cm, 20-30 cm and 0-30 cm for all sites; n = 4 for each attribute at each site.

Site	Soil Attributes	0-10			10-20			20-30			0-30		
		Mean ± SD	Min	Max	Mean ± SD	Min	Max	Mean ± SD	Min	Max	Mean ± SD	Min	Max
G0	BD	0.85 ± 0.11 ^{a}	0.77	1.01	1.10 ± 0.12 ^{a}	0.93	1.22	1.24 ± 0.09 ^{a}	1.17	1.37	1.06 ± 0.19 ^{a}	0.77	1.37
F12	BD	0.94 ± 0.11 ^{a}	0.81	1.05	1.12 ± 0.10 ^{a}	1.05	1.27	1.17 ± 0.06 ^{a}	1.11	1.25	1.08 ± 0.13 ^{a}	0.81	1.27
F20	BD	0.88 ± 0.06 ^{a}	0.80	0.92	1.03 ± 0.10 ^{a}	0.92	1.15	1.23 ± 0.07 ^{a}	1.13	1.30	1.05 ± 0.17 ^{a}	0.80	1.30
F27	BD	0.93 ± 0.06 ^{a}	0.87	1.00	1.06 ± 0.11 ^{a}	0.90	1.15	1.28 ± 0.10 ^{a}	1.13	1.34	1.09 ± 0.17 ^{a}	0.87	1.34
F40	BD	0.95 ± 0.06 ^{a}	0.90	1.04	1.13 ± 0.10 ^{a}	1.00	1.25	1.22 ± 0.10 ^{a}	1.11	1.34	1.10 ± 0.14 ^{a}	0.90	1.34
F47	BD	0.90 ± 0.07 ^{a}	0.82	1.00	1.10 ± 0.12 ^{a}	1.02	1.28	1.26 ± 0.09 ^{a}	1.20	1.39	1.09 ± 0.17 ^{a}	0.82	1.39
G0	SOC	4.5 ± 0.80 ^{ab}	3.7	5.2	3.0 ± 0.15 ^{c}	2.8	3.1	1.5 ± 0.29 ^{ab}	1.2	1.9	3.0 ± 1.4 ^{a}	1.2	5.2
F12	SOC	4.3 ± 0.24 ^{a}	4.1	4.6	2.5 ± 0.14 ^{b}	2.4	2.7	1.6 ± 0.15 ^{a}	1.4	1.8	2.8 ± 1.2 ^{a}	1.4	4.6
F20	SOC	3.8 ± 0.50 ^{ab}	3.1	4.3	2.2 ± 0.24 ^{ab}	2.0	2.6	1.5 ± 0.25 ^{ab}	1.1	1.7	2.5 ± 1.0 ^{a}	1.1	4.3
F27	SOC	3.4 ± 0.47 ^{ab}	2.8	3.9	2.0 ± 0.08 ^{a}	1.9	2.1	1.1 ± 0.17 ^{b}	0.88	1.3	2.2 ± 1.0 ^{a}	0.88	3.9
F40	SOC	3.4 ± 0.18 ^{b}	3.2	3.6	1.9 ± 0.13 ^{a}	1.8	2.1	1.2 ± 0.20 ^{ab}	1.1	1.4	2.2 ± 1.0 ^{a}	1.1	3.6
F47	SOC	3.8 ± 0.44 ^{ab}	3.2	4.2	2.1 ± 0.45 ^{abc}	1.5	2.5	1.2 ± 0.26 ^{ab}	0.79	1.4	2.4 ± 1.2 ^{a}	0.79	4.2
G0	SOCDvol	38.8 ± 11.0 ^{a}	29.2	51.8	32.7 ± 4.9 ^{a}	25.7	37.2	18.7 ± 3.6 ^{a}	14.1	22.8	90.2 ± 18.0 ^{ab}	70.1	111.8
F12	SOCDesm	36.4 ± 5.5 ^{a}	31.6	43.8	28.7 ± 2.2 ^{ab}	26.2	31.2	21.0 ± 2.4 ^{a}	18.6	24.3	86.2 ± 7.0 ^{a}	79.5	94.1
F20	SOCDesm	31.8 ± 3.1 ^{a}	27.2	34.0	24.3 ± 2.7 ^{b}	20.4	26.4	18.1 ± 3.1 ^{a}	14.2	21.6	74.2 ± 8.5 ^{ab}	61.7	79.7
F27	SOCDesm	28.7 ± 2.4 ^{a}	25.9	31.0	23.3 ± 2.0 ^{b}	20.8	25.1	13.8 ± 1.7 ^{a}	11.8	15.9	65.9 ± 3.1 ^{b}	62.7	69.5
F40	SOCDesm	28.9 ± 3.5 ^{a}	26.0	34.0	22.2 ± 3.0 ^{b}	19.9	26.7	16.1 ± 1.9 ^{a}	14.1	18.6	67.3 ± 6.6 ^{b}	61.3	76.3
F47	SOCDesm	32.5 ± 1.8 ^{a}	30.6	34.9	23.2 ± 3.0 ^{b}	20.4	25.9	14.8 ± 2.5 ^{a}	11.2	17.0	70.6 ± 6.9 ^{ab}	62.2	77.7

^{abc}. Means with different lower cased letters are significantly different among sites (p<0.05).

6.4.2. Soil bulk density

There were no significant differences in bulk densities between sites (p>0.05) (Table 6.3). The bulk density of the 0-10 cm depth ranged from the lowest value of 0.85 ± 0.11 g cm^{-3} at site G0 to the highest of 0.95 ± 0.06 g cm^{-3} at site F40. The 10-20 cm depth ranged from the lowest value of 1.03 ± 0.10 g cm^{-3} at site F20 to 1.13 ± 0.10 g cm^{-3} at site F40. The 20-30 cm bulk density values ranged from the lowest of 1.17 ± 0.06 g cm^{-3} at site F12 to the highest of 1.28 ± 0.10 g cm^{-3} at site F27. The bulk density of each site increased with depth.

6.4.3. Soil organic carbon density

The soil organic carbon density (SOCD) decreased with depth for all sites. The SOCD was highest for G0, decreasing with age to its lowest at F27, after which SOCD began to increase again (Table 6.3 and Figure 6.2). In the 0-10 cm soil depth the largest value of SOCD was 38.8 ± 10.9 Mg C ha^{-1} for site G0. From site G0 there was a decline in SOCD to 31.8 ± 3.1 Mg C ha^{-1} at site F20 and the lowest SOCD of 28.7 ± 2.4 Mg C ha^{-1} was at site F27. The SOCD began to increase with age after site F27, with values rising to 28.9 ± 3.5 Mg C ha^{-1} and 32.5 ± 1.8 Mg C ha^{-1} for sites F40 and F47, respectively. However, there were no significant differences between the SOCDs of depth 0-10 cm of each site.

Site G0 had the highest SOCD of the 10-20 cm soil depth with 32.7 ± 4.9 Mg C ha^{-1}, being significantly larger than sites F20, F27, F40 and F47 (p<0.05). From site G0 there was a sustained decrease to the lowest value of 22.2 ± 3.0 Mg C ha^{-1} at site F40, before a small increase to 23.2 ± 3.0 Mg C ha^{-1} at site F47. Site F12 had the highest SOCD of the 20-30 cm soil depth with 21.0 ± 2.4 Mg C ha^{-1}, slightly larger than the 18.7 ± 3.6 Mg C ha^{-1} of site G0. There was a decrease in SOCD from site F12 to site F27 with the lowest value of 13.8 ± 1.7 Mg C ha^{-1}, before an increase with sites F40 and F47 having SOCDs of 16.1 ± 1.9 Mg C ha^{-1} and 14.8 ± 2.5 Mg C ha^{-1}, respectively. There was no significant difference between the sites in the SOCD of the 20-30 cm soil depth.

For the integrated depth 0-30 cm, site G0 had the highest SOCD of all sites with 90.2 ± 18.0 Mg C ha^{-1}, slightly larger than the 86.2 ± 6.7 Mg C ha^{-1} of site F12. From site F12 there was a significant decline (p<0.05) in SOCD to the lowest value of 65.9 ± 3.1 Mg C ha^{-1} at site F27. After site F27, there was a small increase in SOCD to F47, with a value of 70.6 ± 6.9 Mg C ha^{-1}. The change in SOCD of the 0-30 cm soil depth with the stand age is presented in equation 6.16 (Figure 6.3),

$$C_{tot} = 0.0159x^2 - 1.2574x + 92.822 \text{ , } R^2=0.87 \qquad (6.16)$$

where C_{tot} = SOCD, 0-30 cm (Mg C ha^{-1}); x = age of stand (years).

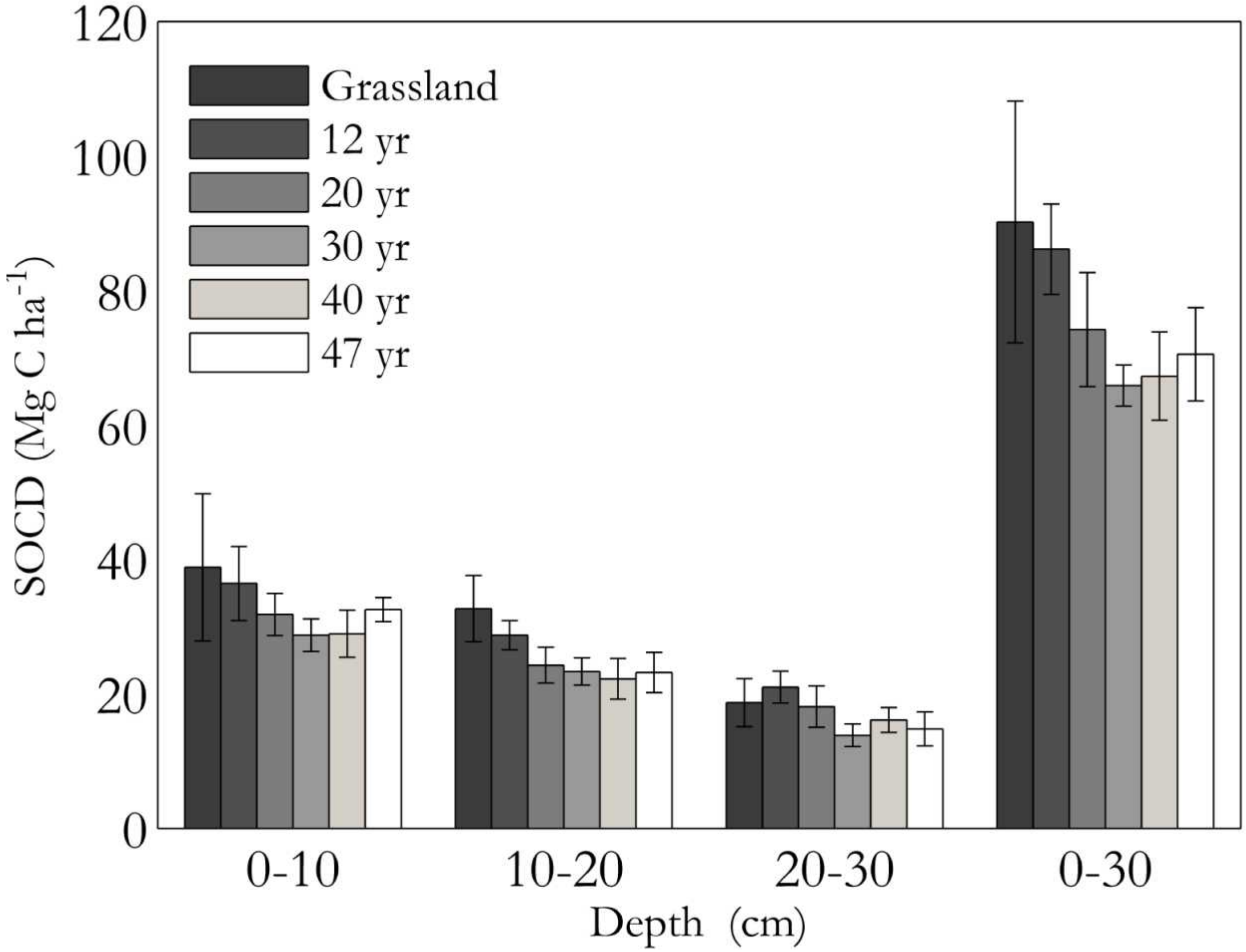

Figure 6.2. Soil organic carbon density (SOCD, Mg C ha^{-1}) versus depth for sites G0, F12, F20, F27, F40 and F47. Error bars indicate standard deviation.

6.4.4. Forest floor and biomass carbon density

The forest floor carbon density (FFCD) increased with the age of the ash stands (Table 6.4). Site F12 had a value of 2.0 ± 0.8 Mg C ha^{-1} and increased to the largest amount of FFCD at site F47 with 8.6 ± 4.6 Mg C ha^{-1}. The values of FFCD were not significantly different between sites (p<0.05). The forest floor consisted mainly of woody debris rather than decayed leaf biomass. The FFCD on average accumulated ~ 0.18 Mg C ha^{-1} yr^{-1} (Figure 6.3). The changes in the FFCD with stand age is presented in equation (6.17),

$$C_{ff} = 0.178x, \quad R^2=0.99 \tag{6.17}$$

where C_{ff} = forest floor carbon density (Mg C ha^{-1}); x = age of stand (years).

The DBH of the ash stands increased with age from 8.7 cm at site F12 to 26.6 cm at site F47. However, the stocking density of the ash stands decreased from 3433 stems ha^{-1} at F12 to 453 stems ha^{-1} at F47 due to thinning activities (Table 6.1). There was an overall increase in the live aboveground biomass carbon density along the chronosequence (Table 6.4). There was a large increase in the live biomass carbon density after afforestation with site F12 having accumulated 41.2 ± 6.6 Mg C ha^{-1}. There was a further increase to 59.8 ± 4.1 Mg C ha^{-1} at site F20, before a decrease in the live biomass carbon density at site F27 with 54.5 ± 3.6 Mg C ha^{-1} as the stocking density almost halves from 1400 stems ha^{-1} at site F20 to 787 stems ha^{-1} at site F27. From site F27 there was a significant increase (p<0.05) to 101.9 ± 5.0 Mg C ha^{-1} at site F40 before another decline to 82.7 ± 0.5 Mg C ha^{-1} again due to the stocking density decrease from 727 (stems ha^{-1}) at site F40 to 453 (stems ha^{-1}) at site F47 (Table 6.1). The biomass carbon density of sites F40 and F47 are significantly larger than sites F12, F20 and F27 (p<0.05). The biomass carbon density of grasslands is much less than that stored in forests. Black et al. (2009) and Peichl et al. (2011a) estimate that between 2.3-2.4 Mg C ha^{-1} is stored in the live biomass carbon density of Irish grasslands.

Dead biomass was noted at only three sites, G0, F20 and F40, and is a very small fraction of the ecosystem carbon density representing only 0.1, 0.3 and 0.1%, respectively of the ecosystem carbon density. The above- and belowground biomass accumulated on average ~ 1.83 Mg C ha^{-1} yr^{-1} (Figure 6.3). The changes in the total above- and belowground carbon density with stand age using a grassland biomass value of 2.35 Mg C ha^{-1} (Black et al., 2009; Peichl et al., 2011a) is presented in equation (6.11),

$$C_b = 1.826x + 12.731 \text{, } R^2\text{=0.86} \qquad (6.18)$$

where C_b = biomass carbon density (Mg C ha^{-1}); x = age of stand (years).

6.4.5. Ecosystem carbon density

The change in ecosystem carbon density over the chronosequence follows the increase in biomass carbon density (Figure 6.3). The ecosystem carbon density increased following afforestation, despite the loss of SOCD, from 90.2 Mg C ha^{-1} at G0 to 161.9 Mg C ha^{-1} at site F47 (Table 6.4). The ecosystem carbon density drops at two sites, F27 and F47 due to the reduction in the biomass carbon density (due to thinning of the stocking numbers). Over the course of the chronosequence, the portion of the ecosystem carbon density stored in the soil decreased from 100% of the ecosystem carbon density, at site G0 to 43.6% of the ecosystem carbon density at site F47 (Figure 6.4). Between sites F27 and F40 the biomass becomes the largest portion of the ecosystem carbon density, larger than the store of SOCD. The changes in the ecosystem carbon density with stand age using a grassland biomass value of 2.35 Mg C ha^{-1} (Black et al., 2009; Peichl et al., 2011a) is presented in equation (6.19),

$$C_e = 1.521x + 100.152 \text{, } R^2\text{=0.81} \qquad (6.19)$$

where C_e = ecosystem carbon density (Mg C ha^{-1}); x = age of stand (years).

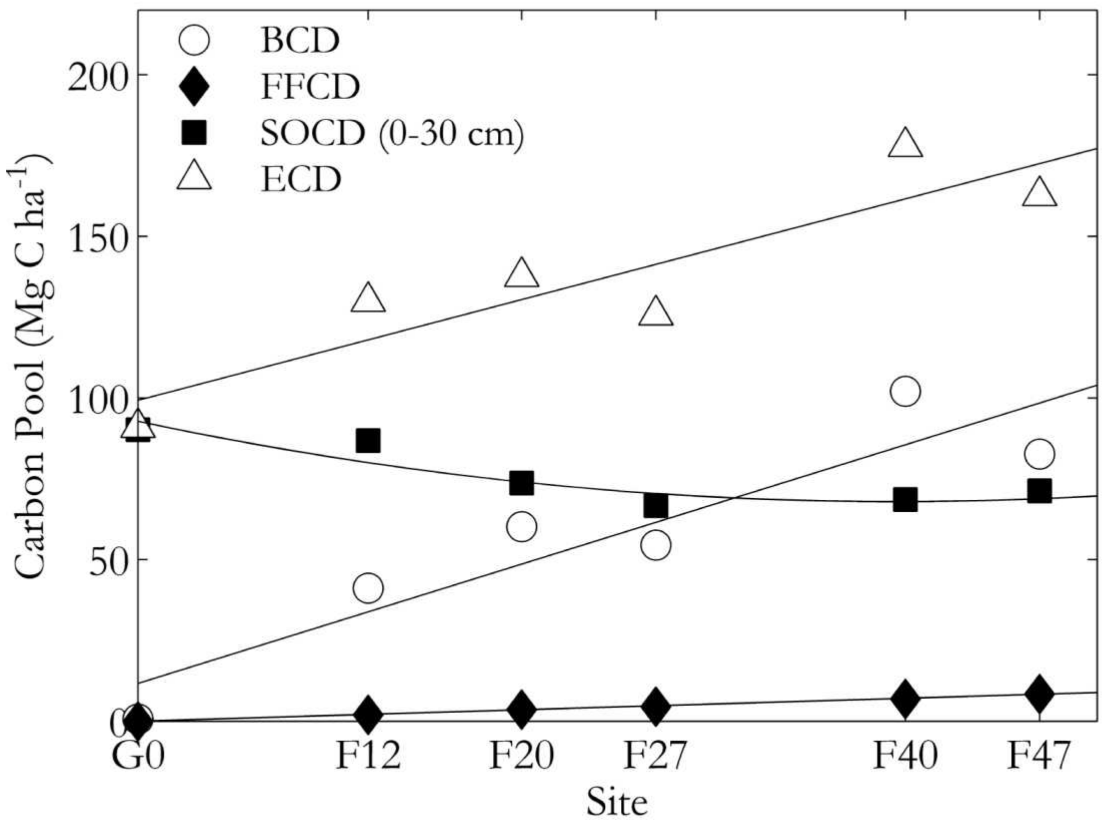

Figure 6.3. Carbon pools (Mg C ha^{-1}) for sites G0, F12, F20, F27, F40 and F47. BCD, biomass carbon density (Mg C ha^{-1}): FFCD, forest floor carbon density (Mg C ha^{-1}): SOCD, soil organic carbon density (Mg C ha^{-1}): ECD, ecosystem carbon density (Mg C ha^{-1}).

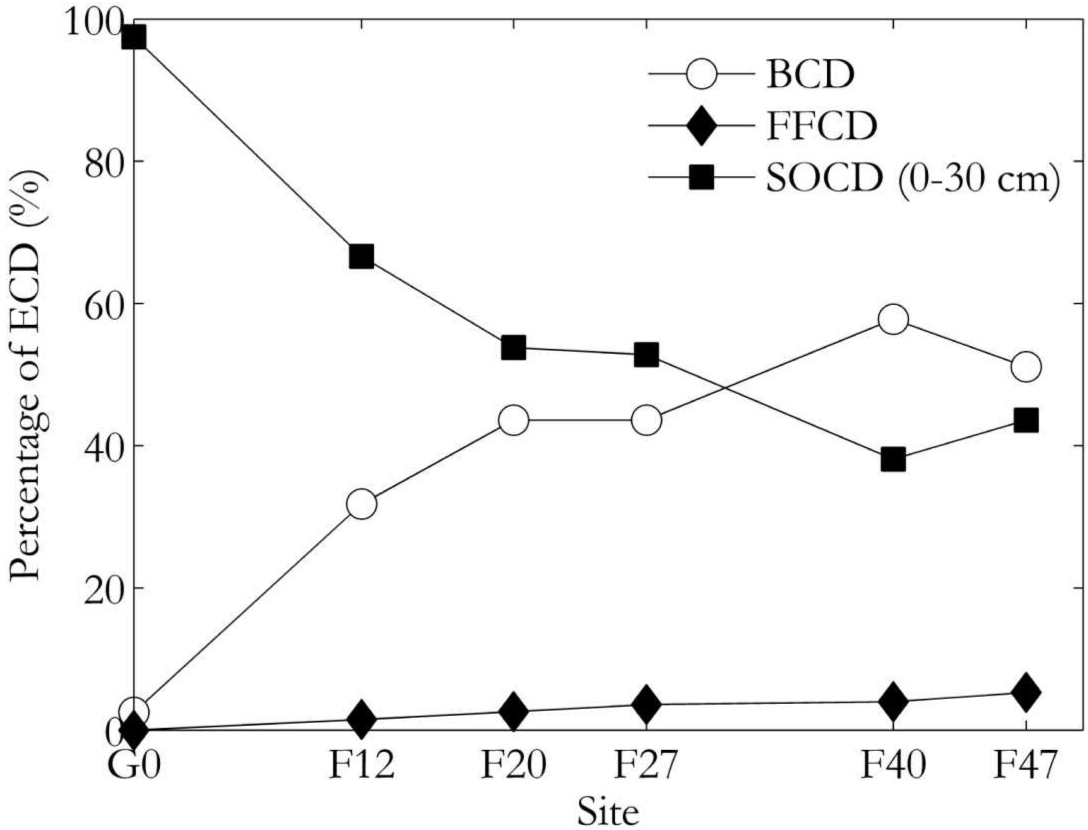

Figure 6.4. Percentage of each carbon pool in relation to the total ecosystem carbon density, for sites G0, F12, F20, F27, F40 and F47. SOCD, soil organic carbon density: FFCD, forest floor carbon density: BCD, biomass carbon density: ECD, ecosystem carbon density.

Table 6.4. The mean ± standard deviation carbon density for each pool (Mg C ha^{-1}) and changes along the chronosequence.

Pool Carbon Density (Mg C ha^{-1})	Site					
	G0	F12	F20	F27	F40	F47
Live Stem & Branch Biomass Carbon Density	-	31.9 ± 5.1	46.5 ± 3.2	42.4 ± 2.8	79.3 ± 3.9	64.4 ± 0.4
Live Leaf Biomass Carbon Density	-	0.8 ± 0.1	1.0 ± 0.1	0.9 ± 0.1	1.6 ± 0.1	1.3 ± 0.03
Live Belowground Biomass Carbon Density	-	8.5 ± 1.4	12.3 ± 0.9	11.2 ± 0.7	21.0 ± 1.0	17.1 ± 0.1
Live Above- & Belowground Biomass Carbon Density	-	41.2 ± 6.6	59.8 ± 4.1	54.5 ± 3.6	101.9 ± 5.0	82.7 ± 0.5
Dead Stem & Branch Biomass Carbon Density	-	-	0.3 ± 0.2	-	0.1 ± 0.2	-
Dead Belowground Biomass Carbon Density	-	-	0.1 ± 0.1	-	0.04 ± 0.06	-
Dead Above- & Belowground Biomass Carbon Density	-	-	0.4 ± 0.3	-	0.2 ± 0.3	-
Forest Floor Carbon Density	-	2.0 ± 0.8	3.6 ± 1.6	4.5 ± 0.7	6.9 ± 6.0	8.4 ± 4.5
Soil Organic Carbon Density (0-30 cm)	90.2 ± 18.0	86.2 ± 7.0	74.2 ± 8.5	65.9 ± 3.1	67.3 ± 6.6	70.6 ± 6.9
Ecosystem Carbon Density	90.2 ± 18.0	129.4 ± 9.7	138.0 ± 9.6	124.9 ± 4.8	176.5 ± 10.3	161.9 ± 8.3

6.5. Discussion

6.5.1. Soil organic carbon

Afforestation of grassland with ash resulted in a loss of soil organic carbon. While the 0-30 cm depth of the grassland had an SOC of 3.0%, the ash stand at age 47 had an SOC of 2.4%, corresponding to a loss of 20% of SOC following afforestation. This occurs as forest establishment initially reduces the SOC of soils as it disturbs soil structure, and leads to an increase in the decomposition of organic matter and thus an increase in C lost to the atmosphere from soil respiration (Zerva and Mencuccini, 2005). Within the first few decades the litter input to the soil from the young forest is small and cannot match the losses of SOC, therefore becoming a source of C (Vesterdal et al., 2002). As the forest ages the organic matter input from plant residues increase and along with reduced soil disturbance the soil switches from losing C to beginning to sequester C (Mao et al., 2010) which occurred at about age 27 in this study.

6.5.2. Soil bulk density

Within the literature it has been noted that the bulk density of soils decreased with age due to the incorporation of organic matter, and so the bulk density decreases while the soil layer becomes thicker (Vesterdal et al., 2002). However, no change in bulk density was observed in this study. This was also noted in Gholz and Fisher (1982), who found no significant differences between the bulk densities in a slash pine chronosequence of 2 to 34 year old stands in Florida, USA.

6.5.3. Soil organic carbon density

Soil organic carbon density follows the pattern of SOC, with a decline following afforestation. This is due to a loss of SOC from forest establishment, which outweighs the organic matter being added to the soil by the young forests. After 27 years, there was an increase in the SOCD of the forest with age, as the inputs of organic matter from the litterfall, woody debris and roots of the forest become greater than the loss of SOCD from decomposition. However, the rate of increase after 27 years is slower than the rate of decline from site G0 to site F27. The increase in SOCD after 27 years is not statistically significant. Over the chronosequence the soil loses C and so as the ash stands reach 60 years (the age that these plantations are scheduled to be harvested) the SOCD is estimated to reach 74.6 Mg C ha^{-1}, representing a 17.3% or 15.6 Mg C ha^{-1} loss from the pre-afforested conditions. This is a large loss and mitigation measures should be considered to limit this SOCD loss. However, more work must be done to assess the SOCD of ash forests older than 47 years to verify this trend. Black et al. (2009) in a chronosequence of Sitka spruce stands on surface-water gley soils in Ireland, found an increase in SOCD, from 97.2 Mg C ha^{-1} at the grassland site to 137.3 Mg C ha^{-1} at the 16 year old spruce stand, an annual increment of 2.2-2.5 Mg C ha^{-1} $year^{-1}$. The authors suggested that this may be due to the higher total belowground C allocation of their sites and reduced decomposition due to the anaerobic conditions associated with the water saturated soils. Tang et al. (2009) in a chronosequence of deciduous forests, predominantly aspen (*Populus tremuloides*) and sugar maple (*Acer saccharum*) after disturbance in Wisconsin and Michigan, USA found an initial decrease in the SOC at 0-60 cm depth of soil from the regenerated stand to the young stand (10 years), and thereafter the SOC increased up to the mature forest (73 years) and the old growth forest (350 years).

The amount of SOCD that is either lost or sequestered depends upon site management, tree species, soil type, length of rotation, age, topography, climate and management (Dixon et al., 1994; Thuille et al., 2000; Jandl et al., 2007). Ash has a short economic rotation length for a broadleaf specie in Ireland at 60-80 years; however, ash is not as productive as the more widely grown conifers, such as Sitka spruce (*Picea sitchensis*). The greater productivity of the conifers is likely to result in a large input of organic matter into the forest floor. However, broadleaf forests have been shown to sequester more C into the soil through a larger root biomass (Laganière et al., 2010).

6.5.4. Biomass carbon density

The significant increase in biomass carbon density (BCD) with the age of the ash stands follows the increase in the mean DBH of the ash trees; however, there are two occasions when BCD declines. These are both due to the decrease in the stocking density of each stand as the sites are thinned. The cutting of trees provides more growing space for the remaining trees (NFI, 2007b).

The biomass carbon density of the forests on average increased by 1.83 Mg C ha^{-1} yr^{-1}, and at the projected time of harvesting, at 60 yrs, the BCD is estimated to be 122.3 Mg C ha^{-1}. The rate that C is sequestered into the BCD of the ash trees is similar to values in the literature. Alberti et al. (2008) in a chronosequence of mixed ash and sycamore in a natural recovery in Italy found that the trees sequestered 1.69 Mg C ha^{-1} yr^{-1}, while Hooker and Compton (2003) in a natural recovery white pine mixed forest in north-eastern, USA, found that the aboveground biomass sequestered 1.53 Mg C ha^{-1} yr^{-1}.

There is uncertainty in the allometric equations used to estimate the biomass of the stands due to their derivation coming from differing locations and species. Equation (6.6) (Bunce, 1968) is an equation based on the stem and branch biomass of ash trees for England, and is the most appropriate allometric equation for Irish ash trees at present, but it does not account for the biomass of the live leaves and the root biomass. Equation (6.7) (Alberti et al., 2005) was based on ash trees from Italy, from a climate different to that of the ash sites sampled within this study. Equation (6.8) (Cairns et al., 1997) is a general measure of the ratio of root biomass to aboveground biomass for a number of species, not just ash. At present these equations represent the best method of estimating the biomass of Irish ash. Future work should focus on producing allometric equations for Irish ash trees that will allow for improved estimates of biomass and biomass carbon density at both the stand and national level.

6.5.5. Ecosystem carbon density

While there was a loss in soil organic carbon density due to afforestation, there was an increase in the biomass carbon density along the chronosequence. As the ash stand grows, the loss of SOCD is more than balanced by ever larger amounts of biomass carbon density, and the ecosystem carbon density follows this trend. At F27 and F47, where the ecosystem carbon density decreases, is due to thinning of the ash stands and the subsequent loss of BCD from the ecosystem as the trees are removed from the sites. At the 40 year old ash stand the biomass becomes a larger store of C than the soil. Over the chronosequence the ecosystem sequesters a mean of 1.52 Mg C ha^{-1} yr^{-1} from the atmosphere, and at 60 years there is an estimated 191.4 Mg C ha^{-1} stored within the ecosystem. Black et al. (2009) in a chronosequence of Sitka spruce stands on surface-water gley soils in Ireland, found that after 10 years the afforested sites become carbon sinks, due to both the increase in soil organic carbon density and the increase in C stored in the above ground biomass increasing from 1.7 Mg C ha^{-1} at the grassland site to 176.5 Mg C ha^{-1} at its maximum at the 45 year old forest. Vesterdal et al. (2002) found a loss in SOC from two separate Oak and Norway spruce forests, however, there was a much greater amount of C stored in the biomass, approximately 75 Mg C ha^{-1} and 105 Mg C ha^{-1} for the Oak and Norway spruce, respectively.

It is Irish government policy to increase the current national forestry area from its current 10% to 17% by 2030, with an aim of 30% of all afforestation being broadleaf species. This study found that by 60 years, afforestation of brown earth grasslands had led to a doubling in ecosystem carbon density, from 92.6 Mg C ha^{-1} at the grassland site to an estimated 191.4 Mg C ha^{-1} at the 60 year old forest. This presents a large C sequestration potential for ash forests within Ireland. However, soils store C for longer than forest biomass, and the loss of SOCD in this study shows that when assessing forest C stocks, soil organic carbon density should always be measured so as to assess the entire ecosystem carbon density. As the forest biomass is normally harvested after 60-80 years, work should be undertaken to assess the changes in ecosystem carbon density following harvesting and reforestation so as to gain insight into impacts of the full rotation of the ash forests. Studies have shown different effects of harvesting on the SOCD. Zerva et al. (2005) found a loss in soil C after deforestation before an increase in soil C in the second rotation of a Sitka spruce chronosequence in England. While other studies have found very little or no change in soil C following harvest (Johnson and Curtis, 2001; Yanai et al., 2003; Martin et al., 2005).

With the recent and continuing afforestation within Ireland, there is a unique opportunity to study the impacts of a number of different tree species on the ecosystem carbon density of a number of different soil types. The chronosequence presented here could be expanded in the next few years, allowing for repetition of the younger sites, and gaining a greater insight into the losses of SOCD from forest establishment. The sampling of the older sites should also be repeated in the next 10-20 years to determine the SOCD of the ash sites before harvesting and post harvest.

This study examined only one soil type, brown earth, which represents ~1/3rd of the total current ash area (NFI, 2007a). The two other major soils that ash is grown on in Ireland are gleys and grey brown podzolics which occupy 27 and 14% respectively. Further work is required to study the effects of ash forests on the SOCD of these two soil types so as to improve the accuracy of the estimates. This is especially the case for the gley soils as soils with high clay contents, >33% (Laganière et al., 2010), such as gleys, have a greater capacity to accumulate SOCD and so may lose less SOCD over time than the brown earth soils presented here.

6.6. Conclusions

There has been very little assessment of the impact of afforestation on ecosystem carbon density in Ireland, especially the impacts of broadleaf afforestation and their effect on soil organic carbon density. We present changes in the carbon pools of ash forests with age following afforestation. We found that the forest SOCD decreased for the first three decades, before beginning to sequester C, but at a slower rate than it was initially lost. However, the SOCD increase is not statistically significant. The C lost from the soil was offset by the increase in the biomass carbon density of the ash forests resulting in the total ecosystem acting as a C sink. Given the Irish governmental policy to increase the national forest cover to 17% by the year 2030, there will be potential to sequester large amounts of C within ash forests as the ecosystem carbon density doubled following afforestation. Future work should assess the impacts of harvesting and reforestation on the ecosystem carbon density in order to fully understand ecosystem C dynamics in afforested ash stands over multiple rotation periods.

6.7. Acknowledgements

This study – Soil Carbon Stock Changes and Greenhouse Gas Flux in Irish Forests (ForestC) - was funded by the Irish Council for Forest Research and Development (COFORD) as part of the Department of Agriculture Fisheries and Food (DAAF) and the by the Science, Technology, Research and Innovation for the Environment (STRIVE) programme 2007 – 2013 (Grant No. 2008-CCRP-1.1A) which is managed by the Irish Environmental Protection Agency (EPA). We would like to thank Dr. Brian Tobin (UCD CARBiFOR II project) for his help in finding and sampling of the sites and the forest managers and owners for permission to sample.

7. Discussion and synthesis

7.1. Paired plot study

Among all paired sites there was a small decrease in soil organic carbon density (0-30 cm) following afforestation. However, when the forest floor carbon density was added to the SOCD, the forest sites became a larger C store than the non-forest site. The paired sites were split into groups between seven different variables that have been shown in the literature to exert an influence on the change in SOCD with afforestation (Guo and Gifford, 2002; Paul et al., 2002; Laganière et al., 2010). We found that the tree species planted can affect SOCD change, with an increase in SOCD under broadleaf species and a decrease under conifer and mixed species (Guo and Gifford, 2002; Laganière et al., 2010; Shi and Cui, 2010). The deeper root system of the broadleaf species produces a larger input of organic matter into the soil from rhizodeposition. We found that the planting of conifer species leads to a decline in SOCD. Conifer species sequester a significantly larger amount of forest floor carbon density than other forest types, and when this is added to the SOCD, it leads to an increase in the carbon stocks of conifer forests from pre-afforestation levels (Paul et al., 2002; Laganière et al., 2010). The change in bulk density values for the conifer forests (an increase) and the broadleaf forests (a decrease) shows that the method used to measure the SOCD of the forest site at different stand ages is very important. The volume based method assumes that bulk density does not change over time, and so the changes in bulk density result in the conifers switching from a loss of SOCD to a gain in SOCD, while the broadleaf forests switch from a gain to a loss of SOCD. This highlights the importance of using the equivalent soil mass (ESM) method in future assessment of SOCD following land use or management changes as it accounts for variation in bulk density (Lee et al., 2009).

We found that afforestation of rough grazing sites and low pH soils may help to minimise the losses of SOCD following afforestation. Rough grazing sites saw the smallest loss in SOCD as the lower net primary productivity of these sites leads to a smaller drop in organic matter inputs following afforestation than for other land uses. A low soil pH can retard soil microbial activity which leads to a drop in SOM decomposition and a possible increase in SOCD due to a longer residence time of new organic matter (van Bergen et al., 1998). We found that sites with larger silt and clay concentrations may sequester more SOCD than other soil textures (Laganière et al., 2010). However, only one site was sampled for each of the silt and clay sites and so no firm conclusions can be drawn from this, but the observation does warrant further research. We found that when analysed by age the change in SOCD decreases up to the 20-29 year group; and thereafter a small increase in the 40+ year group occurs, suggesting that it may take 40+ years for forest soils to sequester enough C to match pre-afforestation levels. The 10-19 year group saw an increase in SOCD, which is contradictory to results noted in the literature.

The variability in the results means that there is no significant change in SOCD for each variable. Further work is required with a larger sample size to verify the findings of this paper and to better understand changes in SOCD following afforestation and their controlling factors.

7.2. Afforested peatlands

The bulk densities of Irish peatlands are within the range reported in the literature (Burke, 1978; Shotbolt et al., 1998; Chapman et al., 2009). The surface peat (0- 20 cm depth) has higher bulk densities, than the deeper peat layers. This is likely due to drainage and subsequent shrinkage of the sites for planting. We found that bulk density does not increase with depth as is widely assumed (Clymo, 1978; Howard et al., 1994; Milne and Brown, 1997; Cruickshank et al., 1998). This finding adds to a number of recent studies that have come to the same conclusion (Tomlinson and Davidson, 2000; Weiss et al., 2002; Lewis et al., 2011). The assumption that bulk density increases with depth may not be appropriate for future peatland modeling as it may lead to an overestimation in the mass of peat, especially in the deeper peats. The mean SOC of the afforested peatland sites are lower (~3%) than values reported within the literature (Tomlinson and Davidson, 2000; Chapman et al., 2009; Lewis et al., 2011), possibly due to the increased decomposition of the catotelm due to lowering of the water table and its subsequent aeration (Minkkinen et al., 2008).

We found that afforested peatlands in Ireland are deeper than had been previously reported with depths of 192 ± 100, 145 ± 130 and 127 ± 100 cm for raised bogs, high level blanket bogs and low level blanket bogs, respectively. These are larger than the peat depths used in the estimation of the national peat C stocks (Tomlinson, 2005). This in turn leads to a possible underestimation of peat C stocks as the soil organic carbon density of the afforested blanket bogs are greater than those used by Tomlinson (2005). This is especially so for the high level blanket bogs in which the soil organic carbon densities estimated in this work are almost three times larger, with a mean SOCD of 774.9 Mg C ha^{-1}. The soil organic carbon density of the high level and low level blanket peat are similar and we suggest that in future they be treated as a single peat type rather than two for future sampling and accounting purpose. The soil organic carbon densities of the raised bogs were significantly larger than those of the blanket bogs as expected. We found depth to be the main predictor of peat soil organic carbon density, and present equations 5.4 and 5.5 as a method to simplify future estimates of afforested peat soil organic carbon density, as the key variable needed is peat depth.

7.3. Ash chronosequence study

Afforestation of grassland with ash leads to large increases in the biomass carbon density, forest floor carbon density and ecosystem carbon density as has been observed in the literature (Vesterdal et al., 2002; Black et al., 2009). However, the SOCD declined significantly following afforestation which is attributed to a decline in SOC concentration. The SOCD declined from the non-forested grassland to the 27 year old stand because the low organic matter input from the young forest cannot match the decomposition of SOM due to soil disturbance during preparation of the site for planting (Vesterdal et al., 2002). As the forest ages the organic matter input to the soil from the aging trees becomes greater than the loss of SOC, and so the soil switches from a source of C to a sink (Turner et al., 2005; Hu et al., 2008). However, the increase in SOCD after 27 years is smaller than the initial decline and we estimate that 60 years after afforestation there is predicted to be a 15.6 Mg C ha^{-1} reduction in SOCD.

The biomass carbon density increased on average by 1.83 Mg C ha^{-1}, which is similar to that found in other studies (Hooker and Compton, 2003; Alberti et al., 2008). The biomass carbon density increased with the DBH of the trees, with two sharp declines due to thinning of the sites at ages 27 and 47. At the 40 year old ash stand the biomass became a larger store of C than the soil. The ecosystem sequesters on average 1.52 Mg C ha^{-1} yr^{-1}. Future ash afforestation may potentially sequester a large amount of C as by 60 years old the ash forest will have over twice as large an ecosystem carbon density than the grassland. However, further work must be done to measure the SOCD of ash stands closer to 60 years to establish the change in ecosystem carbon density over the full rotation.

7.4. Synthesis

Both the paired plot and chronosequence studies noted a decline in SOCD following afforestation of mineral soils, while the peat study noted a smaller SOC compared to the SOC of non-afforested peats in the literature, this decline is attributed to afforestation. However, the paired plot and chronosequence studies show that when other C stores within the forests are measured (forest floor and biomass respectively) the sites show an increase in C compared to the pre-afforestation level.

Although afforestation of peatlands has all but ceased by Coillte, the results of chapter 5 show the importance of peatlands as an SOC store for Ireland. Afforestation can potentially switch peatlands from carbon sources to sinks, and it is important that Coillte continue to not afforest peatlands, and further work done to reduce private peatland afforestation.

The paired plot study found that afforestation with broadleaves, of rough grazing sites or low pH soils may be the most effective management options for foresters to sequester SOCD. However, the chronosequence study found that afforestation with ash leads to a decline in SOCD. This contradiction and the high variance of SOCD show that much research is still required to assess afforestation induced changes within Ireland's forest landscape before any firm conclusions can be made. The data presented here provides a baseline that future studies can resample to measure further changes as the forests age.

8. Conclusions and recommendations

8.1. Conclusions

This study was conducted to assess the current soil organic carbon density (SOCD) of Irish afforested soils and to determine the factors controlling any change. For all sites there was a decline in the SOCD following afforestation, but when other components of the forest ecosystem were also measured afforestation led to an overall sequestration of C. The work of the paired plot study suggests that broadleaf forests are the most effective tree species at sequestering SOCD. However, afforestation of grassland with ash led to a significant loss of C from the soil which was outweighed by a large increase in the biomass carbon density of the forest sites. The paired plot study also suggests that establishing forests on rough grazing sites and on soils with a low pH may also be useful strategies to reduce the loss of SOCD with afforestation.

Chapter 5 is the first major study conducted within Ireland to assess peatland physical properties. Equations 5.4 and 5.5 are presented to improve estimates of afforested peatland soil carbon densities simply by measuring depth, as peat depth accounts for the greatest proportion of variance in soil organic carbon density. The soil organic carbon densities of the high level and low level blanket bogs are very similar and so in future we recommend that for sampling and accounting purposes these sites be analysed simply as blanket bog.

The high variability within all studies means the change in SOCD in Ireland and its controlling factors are far from certain. This work represents a baseline that future studies should use to further examine these questions. With the ongoing afforestation programs being carried-out in Ireland, which is forecast to almost double the national forest area, it is crucial that this work be built upon so as to fully determine the potential sink or source of C that Irish forest soils may be in the future.

8.2. Recommendations for future research

The research conducted under this study has raised some issues and questions that I would like to recommend for future investigations.

The project that this work was part of, ForestC, has a companion study conducted by the CARBiFOR II project, University College Dublin. The CARBiFOR II project sampled 10 peatland sites and 21 paired plot sites selected from the NFI under the same criteria presented in chapter 3.3.1. The 10 peatland sites of CARBiFOR II were incorporated into chapter 5, but the UCD 21 paired sites have yet to be incorporated into chapter 4. The sites will be integrated in a final report for the funding agency, COFORD. The extra paired sites will double the number of sites presented in chapter 4, to 42 sites, and would allow for a greater analysis of the effects of afforestation on mineral soil organic carbon density and the possible controlling factors in Ireland.

This study will be integrated, along with the work of CARBiFOR II, into the CARBWARE model. The CARBWARE model is currently being designed by the CARBWARE project at University College Dublin (Black et al., 2011) to provide a comprehensive analysis of the current total C stocks of the Irish forest state, and to examine how this may change over time and in future climate change scenarios. The CARBWARE model will synthesise the data gathered by ForestC, CARBiFOR II and NFI to meet Ireland's commitments of reporting to the United Nations Framework Convention on Climate Change (UNFCCC).

The CARBWARE project will model the national forest soil C stocks, however, they will not be integrated with the soil C stocks of other land uses. As this study represents the most thorough analysis of forested soil organic carbon density undertaken in Ireland to date, it could be used to improve current estimates of the national soil C stock (Tomlinson, 2005; Eaton et al., 2008; Xu et al., 2011). Current national soil C models have limited data on the SOC, bulk density and SOCD of Irish forest soils, therefore integrating the data of this study would improve current estimates.

During the course of this study it became apparent that there is a lack of data on the soil organic carbon density of peatlands, not only in Ireland, but globally. Chapter 5 provides an analysis of the soil organic carbon densities of Irish afforested peatlands, as well as a methodology appropriate to measure the soil organic carbon density of all peat types and land uses. As peatlands are a large C store, estimated at 53% of the national soil C stock (Tomlinson, 2005) and have the potential to switch from C sinks to sources under climate change, it is imperative to determine the C stocks of non-forested peatlands (Limpens et al., 2008). Any extra sampling could easily be combined with the afforested peat data presented in this work. This combination along with improved measurements of the spatial extent of peat (Connolly et al., 2007; Connolly and Holden, 2009) has the potential to provide a comprehensive estimate of Irish peatland C stocks.

The mineral soils presented within this study should be resampled in the coming years to create a real time dataset to improve the findings of this study. One of the major drawbacks to using the paired site and chronosequence methods is the potential for error based on the assumptions of similarities between the sampled sites. Resampling of these sites would remove these errors. This would also improve the paired sites in that they would no longer be measuring just one point in time. Furthermore, new work should examine in greater detail the impacts of reforestation or conversion to agricultural land etc. on soil organic carbon density, as the impacts of these actions are still not fully understood.

Resampling of the sites will improve our knowledge of soil organic carbon density changes in the later stages of the forests rotation. Forests aged less than 15 years were excluded from analysis in the paired plot and peatland studies, leaving a 15 year gap in the earliest stages of forest growth that was not measured. To amend this knowledge gap, it is suggested that additional resampling sites should be identified and sampled in coming years. It is Irish government policy to increase afforestation from 10% of land cover in 2007 (NFI, 2007a) to 17% by the year 2030 (Department of Agriculture, 1996). This high rate of afforestation provides a unique opportunity to set up a large number of resampling studies. Sites should be identified and sampled prior to site cultivation and planting so as to fully measure the impact of site cultivation and afforestation on soil organic carbon density in the earliest stages of the forest rotation.

Future resampling efforts should increase the soil sampling depth to at least 50 cm. As the forests age, the root biomass of the trees becomes larger and penetrates deeper, potentially storing C in deeper parts of the soil profile, especially broadleaf species. The mineral soil was measured to 30 cm in this study, and so may be underestimating the C sequestration ability of forests as it does not measure changes in the deeper soil.

Study of the growth of tree biomass in Ireland has so far focused primarily on Sitka spruce (Saiz et al., 2006; Tobin et al., 2006), while the growth of young ash (*Fraxinus excelsior* L.) and black alder (*Alnus glutinosa L.* Peichl et al., 2011b) has also been measured, but less widely. Along with resampling of the soil and forest floor carbon density as mentioned above, it would also be prudent to produce further allometric data and models for both broadleaf and conifer tree species planted in Ireland, such as Norway spruce, lodgepole pine, Japanese larch, Douglas fir, oak, sycamore etc. This would vastly improve the estimates of biomass carbon density changes in Irish forests with age and improve the estimates of the total C stored in the Irish forest biomass.

The mineral and peat sites sampled within this study were selected from the Irish national forest inventory (NFI). The original NFI was sampled between 2004 - 2006 and published in 2007 (NFI, 2007a). In 2010, the NFI began to resample the sites as part of a continuous 5 year resampling program. Within the NFI methodology a soil pit is dug and the soil type is recorded, but no samples are taken (NFI, 2007b). To improve measurements of forest SOCD, samples could be taken at each NFI site to a set soil depth as part of the NFI measurements. This would provide a soil database of 1,742 forest sites that would be resampled every 5 years for SOC, and would support the suggested resampling regimes with a higher sampling density that could be established. Due to the commitment of the NFI to measure a number of properties at each forest site, it should be a small soil sampling regime, so as to fit into the NFI dataset. This has been successfully implemented in the Italian NFI where soil sampling was integrated into the existing biomass measurements (M. Rodeghiero, personal communication).

To date in Ireland there have been few greenhouse gas (GHG) flux studies on afforested sites over a sustained time period. Peichl et al. (2011b) observed an initial decrease in the gross and net ecosystem production and ecosystem respiration following recent afforestation of grassland with ash and alder in Dripsey, County Cork. Mishurov and Kiely (2010) at the same mixed ash and alder site recorded an increase in N_2O emissions following cultivation of the site, before a decrease over the next three years to levels that were ~one third of the emissions of surrounding grasslands. Another study by Black et al. (2007) measured the CO_2 fluxes of a 14 year old Sitka spruce forest in the Irish midlands, measuring an uptake of 7.30-11.44 Mg C ha^{-1} using both ecological inventory and eddy covariance methods. Nieuwenhuis et al. (2011) are currently measuring the CO_2 fluxes of an ash afforestation chronosequence. Eddy covariance is a micrometeorological technique that measures the turbulent flux across the vegetation canopy-atmosphere layer to determine the net difference of material moving across this interface (Lenschow, 1995; Baldocchi, 2003; Sottocornola, 2007). An eddy covariance system could be set up prior to site cultivation and planting to measure the CO_2, N_2O and CH_4 fluxes of the forest during planting and as the forests grow. This would create a high quality dataset of the total ecosystem flux of Irish forests, however, eddy covariance systems are expensive, and would require constant maintenance.

9. References

Aciego Pietri, J.C., Brookes, P.C., 2008. Relationships between soil pH and microbial properties in a UK arable soil. Soil Biology and Biochemistry. 40, 1856–1861.

Ågren, G.I., Bosatta, E., Magill, A.H., 2001. Combining theory and experiment to understand effects of inorganic nitrogen on litter decomposition. Oecologia. 128, 94–98.

Alberti, G., Candido, P., Peressotti, A., Turco, S., Piussi, P., Zerbi, G., 2005. Aboveground biomass relationships for mixed ash (*Fraxinus excelsior* L. and *Ulmus glabra* Hudson) stands in Eastern Prealps of Friuli Venezia Giulia (Italy). Annals of Forest Science. 62, 831 - 836.

Alberti, G., Peressotti, A., Piussi, P., Zerbi, G., 2008. Forest ecosystem carbon accumulation during a secondary succession in the Eastern Prealps of Italy. Forestry. 81, 1 -11.

Allison, V.J., Miller, R.M., Jastrow, J.D., Matamala, R., Zak, D.R., 2005. Changes in soil microbial community structure in a tallgrass prairie chronosequence. Soil Science Society of America Journal. 69, 1412–1421.

Anderson, D.E., 2002. Carbon accumulation and C/N ratios of peat bogs in North-West Scotland. Scottish Geographical Journal. 118, 323-341.

Anderson, R.A., Pyatt, D.G., Sayers, J.M., Blackhall, S.R., Robinson, H.D., 1992. Volume and mass budgets of blanket peat in the north of Scotland. Suo. 43, 195-198.

Andersson, S., Nilsson, S.I., 2001. Influence of pH and temperature on microbial activity, substrate availability of soil-solution bacteria and leaching of dissolved organic carbon in a mor humus. Soil Biology and Biochemistry. 33, 1181–1191.

Augusto, L., Ranger, J., Binkley, D., Rothe, A., 2002. Impact of several common tree species of European temperate forests on soil fertility. Annals of Forest Science. 59, 233–253.

Baer, S.G., Kitchen, D.J., Blair, J.M., Rice, C.W., 2002. Changes in ecosystem structure and function along a chronosequence of restored grasslands. Ecological Applications. 12, 1688–1701.

Baldocchi, D.D., 2003. Assessing the eddy covariance technique for evaluating carbon dioxide exchange rates of ecosystems: past, present and future. Global Change Biology. 9, 479–492.

Banning, N.C., Grant, C.D., Jones, D.L., Murphy, D.V., 2008. Recovery of soil organic matter, organic matter turnover and nitrogen cycling in a post-mining forest rehabilitation chronosequence. Soil Biology and Biochemistry. 40, 2021–2031.

Bashkin, M.A., Binkley, D., 1998. Changes in soil carbon following afforestation in Hawaii. Ecology. 79, 828–833.

Batjes, N.H., 1996. Total carbon and nitrogen in the soils of the world. European Journal of Soil Science. 47, 151-163.

Binkley, D., Resh, S.C., 1999. Rapid changes in soils following Eucalyptus afforestation in Hawaii. Soil Science Society of America Journal. 63, 222-225.

Biswas, T., Mukherjee, S., 1995. Textbook of Soil Sciences. McGraw-Hill Inc., USA.

Black, K., Bolger, T., Davis, P., Nieuwenhuis, M., Reidy, B., Saiz, G., Tobin, B., Osborne, B., 2007. Inventory and eddy covariance-based estimates of annual carbon sequestration in a Sitka spruce (*Picea sitchensis* (Bong.) Carr.) forest ecosystem. European Journal of Forest Research. 126, 167–178.

Black, K., Byrne, K.A., Mencuccini, M., Tobin, B., Nieuwenhuis, M., Reidy, B., Bolger, T., Saiz, G., Green, C., Farrell, E.T., Osborne, B., 2009. Carbon stock and stock changes across a *Sitka spruce* chronosequence on surface-water gley soils. Forestry. 82, 255-272.

Black, K., Gallagher, G., Connelly, J., Hawkins, M., Tarleton, M., Hamel, J., 2011. CARBWARE. http://www.coford.ie/researchprogramme/thematicareapolicyandpublicgoods/forestsandclimatechange/carbware/.

Bock, M.D., Van Rees, K.C.J., 2002. Mechanical site preparation impacts on soil properties and vegetation communities in the Northwest Territories. Canadian Journal of Forest Research. 32, 1381–1392.

Bolinder, M.A., Angers, D.A., Bélanger, G., Michaud, R., Laverdière, M.R., 2002. Root biomass and shoot to root ratios of perennial forage crops in eastern Canada. Canadian Journal of Plant Science. 82, 731–737.

Bond-Lamberty, B., Wang, C., Gower, S.T., 2004. Net primary production and net ecosystem production of a boreal black spruce wildfire chronosequence. Global Change Biology. 10, 473–487.

Braekke, F.H., Finér, L., 1991. Fertilization effects on surface peat of pine bogs. Scandinavian Journal of Forest Research. 6, 443-449.

Breuer, L., Huisman, J.A., Keller, T., Frede, H.-G., 2006. Impact of a conversion from cropland to grassland on C and N storage and related soil properties: Analysis of a 60-year chronosequence. Geoderma. 133, 6-18.

Bridgham, S.D., Richardson, C.J., Maltby, E., Faulkner, S.P., 1991. Cellulose decay in natural and disturbed peatlands in North-Carolina. Journal of Environmental Quality. 20, 695-701.

Bubier, J.L., Moore, T.R., Bellisario, L., Comer, N.T., Crill, P.M., 1995. Ecological controls on methane emissions from a northern peatland complex in the zone of discontinuous permafrost, Manitoba, Canada. Global Biogeochemical Cycles. 9, 455–470.

Bunce, R.G.H., 1968. Biomass and production of trees in a mixed deciduous woodland: I. Girth and height as parameters for the estimation of tree dry weight. Journal of Ecology. 56, 759-775.

Burke, W., 1978. Long-term effects of drainage and land use on some physical properties of blanket peat. Irish Journal of Agricultural Research. 17, 315-322.

Burnham, J.H., Sletten, R.S., 2010. Spatial distribution of soil organic carbon in northwest Greenland and underestimates of high Arctic carbon stores. Global Biogeochemical Cycles. 24, GB3012, doi:10.1029/2009GB003660.

Byrne, K.A., Farrell, E.P., 1997. The influence of forestry on blanket peatland. in: Hayes, M.H.B., Wilson, W.S., (Eds.), Humic Substances , Peats and Sludges: Health and Environmental Aspects. The Royal Society of Chemistry. pp. 262-277.

Byrne, K.A., Farrell, E.P., 2005. The effect of afforestation on soil carbon dioxide emissions in blanket peatland in Ireland. Forestry. 78, 217 -227.

Byrne, K.A., Milne, R., 2006. Carbon stocks and sequestration in plantation forests in the Republic of Ireland. Forestry. 79, 361-369.

Cairns, M.A., Brown, S., Helmer, E.H., Baumgardner, G.A., 1997. Root biomass allocation in the world's upland forests. Oecologia. 111, 1-11.

Carter, M.R., Gregorich, E.G., Angers, D.A., Donald, R.G., Bolinder, M.A., 1998. Organic C and N storage, and organic C fractions, in adjacent cultivated and forested soils of eastern Canada. Soil and Tillage Research. 47, 253–261.

Cerri, C.C., Volkoff, B., Andreaux, F., 1991. Nature and behaviour of organic matter in soils under natural forest, and after deforestation, burning and cultivation, near Manaus. Forest Ecology and Management. 38, 247–257.

Chapman, S.J., Bell, J., Donnelly, D., Lilly, A., 2009. Carbon stocks in Scottish peatlands. Soil Use and Management. 25, 105-112.

Chen, C.R., Condron, L.M., Davis, M.R., Sherlock, R.R., 2000. Effects of afforestation on phosphorus dynamics and biological properties in a New Zealand grassland soil. Plant and Soil. 220, 151-163.

Chen, C., Xu, Z., Mathers, N., 2004. Soil carbon pools in adjacent natural and plantation forests of subtropical Australia. Soil Science Society of America Journal. 68, 282-291.

Chmielewski, K., 1991. The effect of habitat conditions on microbiological activity of peat soils. Polish Ecological Studies. 17, 143-153.

Clymo, R.S., 1978. A model of peat bog growth. in: Heal, O.W., Perkins, D.F. (Eds.), Production Ecology of British Moors and Montane Grasslands. Springer, Berlin, Germany. pp. 187-223.

Conen, F., Zerva, A., Arrouays, D., Jolivet, C., Jarvis, P.G., Grace, J., Mencuccini, M,. 2005. The carbon balance of forests soils: detectability of changes in soil carbon stocks in temperate and boreal forests. in: Griffith, H. Jarvis, P.G,. (Eds.), The Carbon Balance of Forest Biomes. Bios Scientific, pp. 233-247.

Connolly, J., Holden, N.M., Ward, S.M., 2007. Mapping peatlands in Ireland using a rule-based methodology and digital data. Soil Science Society of America Journal. 71, 492-499.

Connolly, J., Holden, N., 2009. Mapping peat soils in Ireland: updating the derived Irish peat map. Irish Geography. 42, 343-352.

Corre, M.D., Schnabel, R.R., Shaffer, J.A., 1999. Evaluation of soil organic carbon under forests, cool-season and warm-season grasses in the northeastern US. Soil Biology and Biochemistry. 31, 1531-1539.

Cruickshank, M.M., Tomlinson, R.W., Devine, P.M., Milne, R.M., 1998. Carbon in the vegetation and soils of Northern Ireland. Biology and Environment - Proceedings of the Royal Irish Academy. 98B, 9-21.

Cruickshank, M.M., Tomlinson, R.W., Trew, S., 2000. Application of CORINE land-cover mapping to estimate carbon stored in the vegetation of Ireland. Journal of Environmental Management. 58, 269–287.

Davis, M., 2001. Soil properties under pine forest and pasture at two hill country sites in Canterbury. New Zealand Journal of Forestry Science. 31, 3-17.

Davis, M.R., Condron, L.M., 2002. Impact of grassland afforestation on soil carbon in New Zealand: a review of paired-site studies. Australian Journal of Soil Research. 40, 675-690.

Davis, M.R., Allen, R.B., Clinton, P.W., 2003. Carbon storage along a stand development sequence in a New Zealand Nothofagus forest. Forest Ecology and Management. 177, 313-321.

Davis, M.R., Wilde, R.H., Garrett, L., Oliver, G., 2004. New Zealand Carbon Monitoring System Soil Data Collection Manual. Prepared for: New Zealand Climate Change Office Ministry for the Environment, PO Box 10-362, Wellington.

Davis, M., Nordmeyer, A., Henley, D., Watt, M., 2007. Ecosystem carbon accretion 10 years after afforestation of depleted subhumid grassland planted with three densities of *Pinus nigra*. Global Change Biology. 13, 1414–1422.

DeGryze, S., Six, J., Paustian, K., Morris, S.J., Paul, E.A., Merckx, R., 2004. Soil organic carbon pool changes following land-use conversions. Global Change Biology. 10, 1120-1132.

Del Galdo, I., Six, J., Peressotti, A., Cotrufo, M.F., 2003. Assessing the impact of land-use change on soil C sequestration in agricultural soils by means of organic matter fractionation and stable C isotopes. Global Change Biology. 9, 1204–1213.

Department of Agriculture, 1996. Growing for the Future - A Strategic Plan for the Development of the Forestry Sector in Ireland. Department of Agriculture, Food and Forestry, The Stationery Office, Dublin, Ireland.

Dixon, R.K., Brown, S., Houghton, R.A., Solomon, A.M., Trexler, M.C., Wisniewski, J., 1994. Carbon pools and flux of global forest ecosystems. Science. 263, 185–189.

Eaton, J.M., 2007. A review of the current knowledge of soil carbon stocks and stock changes in the Republic of Ireland. Unpublished Work.

Eaton, J.M., McGoff, N.M., Byrne, K.A., Leahy, P., Kiely, G., 2008. Land cover change and soil organic carbon stocks in the Republic of Ireland 1851–2000. Climatic Change. 91, 317-334.

Ellert, B.H., Bettany, J.R., 1995. Calculation of organic matter and nutrients stored in soils under contrasting management regimes. Canadian Journal of Soil Science. 75, 529-538.

Ellert, B.H., Janzen, H.H., McConkey, B.G., 2001. Measuring and comparing soil carbon storage. in: Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A. (Eds.), Assessment Methods for Soil Carbon. Lewis Publishers, Boca Raton, FL, pp. 131–144.

Eswaran, H., Reich, P.F., Kimble, J.M., Beinroth, F.H., Padmanabhan, E., Moncharoen, P., 2000. Global carbon stocks, in: Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A. (Eds.), Global Climate Change and Pedogenic Carbonates. Lewis Publishers, Boca Raton, FL, pp. 15–25.

Falloon, P.D., Smith, P., 2000. Modelling refractory soil organic matter. Biology and Fertility of Soils. 30, 388–398.

Farley, K.A., Kelly, E.F., Hofstede, R.G.M., 2004. Soil organic carbon and water retention after conversion of grasslands to pine plantations in the Ecuadorian Andes. Ecosystems. 7, 729–739.

Foss, P.J., O'Connell, C.A., Crushell, P.A., 2001. Bogs and Fens of Ireland Conservation Plan 2005. Irish Peatland Conservation Council, Dublin, Ireland.

Franklin, O., Högberg, P., Ekblad, A., Ågren, G.I., 2003. Pine forest floor carbon accumulation in response to N and PK additions: bomb 14 C modelling and respiration studies. Ecosystems. 6, 644–658.

Franzluebbers, A.J., Stuedemann, J.A., Schomberg, H.H., Wilkinson, S.R., 2000. Soil organic C and N pools under long-term pasture management in the Southern Piedmont USA. Soil Biology and Biochemistry. 32, 469–478.

Freeman, C., Ostle, N., Kang, H., 2001. An enzymic "latch" on a global carbon store. Nature. 409, 149.

Gardiner, M.J., Radford, T., 1980. Soil associations of Ireland and their land use potential. Explanatory bulletin to soil map of Ireland. Soil survey bulletin no 36. An Foras Talúntais, Dublin, Ireland.

Garten Jr, C.T., Ashwood, T.L., 2002. Landscape level differences in soil carbon and nitrogen: Implications for soil carbon sequestration. Global Biogeochemical Cycles. 16, 1114.

Garten, C.T., Wullschleger, S.D., 1999. Soil carbon inventories under a bioenergy crop (switchgrass): Measurement limitations. Journal of Environmental Quality. 28, 1359–1365.

Gholz, H.L., Fisher, R.F., 1982. Organic matter production and distribution in slash pine (*Pinus elliottii*) plantations. Ecology. 63, 1827-1839.

Gifford, R.M., Roderick, M.L., 2003. Soil carbon stocks and bulk density: spatial or cumulative mass coordinates as a basis of expression? Global Change Biology. 9, 1507-1514.

Goidts, E., Van Wesemael, B., Crucifix, M., 2009. Magnitude and sources of uncertainties in soil organic carbon (SOC) stock assessments at various scales. European Journal of Soil Science. 60, 723–739.

Grünzweig, J.M., Gelfand, I., Yakir, D., 2007. Biogeochemical factors contributing to enhanced carbon storage following afforestation of a semi-arid shrubland. Biogeosciences Discussions. 4, 2111–2145.

Guo, L.B., Wang, M., Gifford, R.M., 2007. The change of soil carbon stocks and fine root dynamics after land use change from a native pasture to a pine plantation. Plant and Soil. 299, 251–262.

Guo, L.B., Gifford, R.M., 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biology. 8, 345-360.

Hammond, R.F., 1981. The Peatlands of Ireland. An Foras Talúntais, Dublin, Ireland.

Hassink, J., 1997. The capacity of soils to preserve organic C and N by their association with clay and silt particles. Plant and Soil. 191, 77–87.

Hedde, M., Aubert, M., Decaëns, T., Bureau, F., 2008. Dynamics of soil carbon in a beechwood chronosequence forest. Forest Ecology and Management. 255, 193-202.

Hoegberg, P., Fan, H., Quist, M., Binkley, D., Tamm, C.O., 2006. Tree growth and soil acidification in response to 30 years of experimental nitrogen loading on boreal forest. Global Change Biology. 12, 489–499.

Holden, J., Burt, T.P., 2002. Piping and pipeflow in a deep peat catchment. CATENA. 48, 163-199.

Holden, J., 2006. Sediment and particulate carbon removal by pipe erosion increase over time in blanket peatlands as a consequence of land drainage. Journal of Geophysical Research. 111, F02010.

Holden, J., Burt, T.P., Vilas, M., 2002. Application of ground-penetrating radar to the identification of subsurface piping in blanket peat. Earth Surface Processes and Landforms. 27, 235-249.

Hooker, T.D., Compton, J.E., 2003. Forest ecosystem carbon and nitrogen accumulation during the first century after agricultural abandonment. Ecological Applications. 13, 299-313.

Horgan, T., Keane, M., McCarthy, R., Lally, M., Thompson, D., 2004. Guide to forest tree species selection and silviculture in Ireland, second. ed. COFORD, National Council for Forest Research and Development. Dublin, Ireland.

Howard, P.J.A., Loveland, P.J., Bradley, R.I., Dry, F.T., Howard, D.M., Howard, D.C., 1994. The carbon content of soil and its geographical distribution in Great Britain. Soil Use and Management. 11, 9-15.

Hu, Y.L., Zeng, D.H., Fan, Z.P., Chen, G.S., Zhao, Q., Pepper, D., 2008. Changes in ecosystem carbon stocks following grassland afforestation of semiarid sandy soil in the southeastern Keerqin Sandy Lands, China. Journal of Arid Environments. 72, 2193–2200.

Huang, M., Ji, J., Li, K., Liu, Y., Yang, F., Tao, B., 2007. The ecosystem carbon accumulation after conversion of grasslands to pine plantations in subtropical red soil of South China. Tellus B. 59, 439–448.

Insam, H., Domsch, K.H., 1988. Relationship between soil organic carbon and microbial biomass on chronosequences of reclamation sites. Microbial Ecology. 15, 177–188.

IPCC, 1997. Revised Guidelines for National Greenhouse Gas Inventories. Intergovernmental Panel on Climate Change. Organization for Economic Co-operation (OECD) and International Energy Agency.

IPCC, 2007. The Physical Science Basis, Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, edited by: Solomon, S., Qin, D., Manning, M., Chen, Z., Arquis, M., Averyt, K. B., Tignor, M., and Miller, H. L., Cambridge University Press, Cambridge, UK and New York, USA, pp. 996, http://www.ipcc.ch/ipccreports/ar4-wg1.html.

Jackson, R.B., Banner, J.L., Jobbágy, E.G., Pockman, W.T., Wall, D.H., 2002. Ecosystem carbon loss with woody plant invasion of grasslands. Nature. 418, 623–626.

Jandl, R., Lindner, M., Vesterdal, L., Bauwens, B., Baritz, R., Hagedorn, F., Johnson, D.W., Minkkinen, K., Byrne, K.A., 2007. How strongly can forest management influence soil carbon sequestration? Geoderma. 137, 253-268.

Jobbágy, E.G., Jackson, R.B., 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological Applications. 10, 423-436.

Johansson, M.B., 1994. The influence of soil scarification on the turn-over rate of slash needles and nutrient release. Scandinavian Journal of Forest Research. 9, 170–179.

Johnson, D.W., 1992. Effects of forest management on soil carbon storage. Water, Air, & Soil Pollution. 64, 83-120.

Johnson, D.W., Curtis, P.S., 2001. Effects of forest management on soil C and N storage: meta analysis. Forest Ecology and Management. 140, 227-238.

Johnson, E.A., Miyanishi, K., 2008. Testing the assumptions of chronosequences in succession. Ecology Letters. 11, 419–431.

Joyce, P.M., Huss, J., McCarthy, R., Pfeifer, A., Hendrick, E., 1998. Growing Broadleaves. Silvicultural Guidelines for Ash, Sycamore, Wild cherry, Beech and Oak in Ireland. COFORD, National Council for Forest Research and Development. Dublin, Ireland.

Jug, A., Makeschin, F., Rehfuess, K.E., Hofmann-Schielle, C., 1999. Short-rotation plantations of balsam poplars, aspen and willows on former arable land in the Federal Republic of Germany. III. Soil ecological effects. Forest Ecology and Management. 121, 85-99.

Kashian, D.M., Turner, M.G., Romme, W.H., 2005. Variability in leaf area and stemwood increment along a 300-year lodgepole pine chronosequence. Ecosystems. 8, 48–61.

Kechavarzi, C., Dawson, Q., Leeds-Harrison, P.B., 2010. Physical properties of low-lying agricultural peat soils in England. Geoderma. 154, 196-202.

Kiely, G., McGoff, N., Eaton, J., Leahy, P., Xu, X., Carton, O., 2010. SoilC - Measuring and Modelling of Soil Carbon Stocks and Stock Changes in Irish Soils. EPA report Strive No.35 ISBN: 978-1-84995-320-6.

Kirschbaum, M.U.F., Guo, L.B., Gifford, R.M., 2008. Why does rainfall affect the trend in soil carbon after converting pastures to forests? A possible explanation based on nitrogen dynamics. Forest Ecology and Management. 255, 2990–3000.

Koehler, A.-K., 2010. The Carbon Balance of a Blanket Peat Catchment. Ph.D. Thesis, Department of Civil and Environmental Engineering, University College Cork, Cork, Ireland.

Koehler, A.-K., Sottocornola, M., Kiely, G., 2010. How strong is the current carbon sequestration of an Atlantic blanket bog? Global Change Biology. 17, 309-319.

Kuzyakov, Y., Domanski, G., 2000. Carbon input by plants into the soil. Review. Journal of Plant Nutrition and Soil Science. 163, 421–431.

Kyoto Protocol, 1997. Kyoto Protocol to the United Nations Framework Convention on Climate Change. FCCC/CP/1997/7/Add.1, Decision 1/CP.3, Annex 7. UN.

Laclau, P., 2003. Biomass and carbon sequestration of ponderosa pine plantations and native cypress forests in northwest Patagonia. Forest Ecology and Management. 180, 317–333.

Laganière, J., Angers, D.A., Parè, D., 2010. Carbon accumulation in agricultural soils after afforestation: a meta-analysis. Global Change Biology. 16, 439-453.

Lal, R., 2004. Soil carbon sequestration to mitigate climate change. Geoderma. 123, 1–22.

Lal, R., 2005. Forest soils and carbon sequestration. Forest Ecology and Management. 220, 242–258.

Lee, J., Hopmans, J.W., Rolston, D.E., Baer, S.G., Six, J., 2009. Determining soil carbon stock changes: Simple bulk density corrections fail. Agriculture, Ecosystems & Environment. 134, 251-256.

Lemma, B., Kleja, D.B., Nilsson, I., Olsson, M., 2006. Soil carbon sequestration under different exotic tree species in the southwestern highlands of Ethiopia. Geoderma. 136, 886–898.

Lenschow, D.H., 1995. Micrometeorological techniques for measuring biosphere-atmosphere trace gas exchange, in: Matson, P. A. and Harriss, R. C. (Eds.) Biogenic Trace Gases: Measuring Emissions From Soil and Water, pp. 126-163. Blackwell Science, London, UK.

Lewis, C., Albertson, J., Xu, X., Kiely, G., 2011. Spatial variability of hydraulic conductivity and bulk density along a blanket peatland hillslope. Unpublished Work.

Lieffers, V.J., 1988. Sphagnum and cellulose decomposition in drained and natural areas of an Alberta peatland. Canadian Journal of Soil Science. 68, 755-761.

Lima, A., Silva, I.R., Neves, J.C.L., Novais, R.F., Barros, N.F., Mendonça, E.S., Smyth, T.J., Moreira, M.S., Leite, F.P., 2006. Soil organic carbon dynamics following afforestation of degraded pastures with eucalyptus in southeastern Brazil. Forest Ecology and Management. 235, 219–231.

Limpens, J., Berendse, F., Blodau, C., Canadell, J.G., Freeman, C., Holden, J., Roulet, N., Rydin, H., Schaepman-Strub, G., 2008. Peatlands and the carbon cycle: from local processes to global implications - a synthesis. Biogeosciences. 5, 1475-1491.

Lindsay, R., 2010. Peatbogs and Carbon: a Critical Synthesis to Inform Policy Development in Oceanic Peat Bog Conservation and Restoration in the Context of Climate Change. RSPB, Scotland.

Lorenz, K., Lal, R., 2005. The depth distribution of soil organic carbon in relation to land use and management and the potential of carbon sequestration in subsoil horizons. Advances in Agronomy. 88, 35–66.

Mallik, A.U., Hu, D., 1997. Soil respiration following site preparation treatments in boreal mixedwood forest. Forest Ecology and Management. 97, 265–275.

Mao, R., Zeng, D.-H., Hu, Y.-L., Li, L.-J., Yang, D., 2010. Soil organic carbon and nitrogen stocks in an age-sequence of poplar stands planted on marginal agricultural land in Northeast China. Plant and Soil. 332, 277-287.

Markewitz, D., Sartori, F., Craft, C., 2002. Soil change and carbon storage in longleaf pine stands planted on marginal agricultural lands. Ecological Applications. 12, 1276-1285.

Martin, J.L., Gower, S.T., Plaut, J., Holmes, B., 2005. Carbon pools in a boreal mixedwood logging chronosequence. Global Change Biology. 11, 1883-1894.

McKnight, T.L., Hess, D., 2007. Physical Geography: A Landscape Appreciation, ninth ed. Prentice Hall, Upper Saddle River, NJ.

McLean, E.O., 1982. Soil pH and lime requirement. in: Page, A.L., Miller, R.H., Keeney, D.R., (Eds), Methods of Soil Analysis, Part 2. Agronomy 9, second ed. ASA and SSSA, Madison, WI, USA.

Mendoza-Ponce, A., Galicia, L., 2010. Aboveground and belowground biomass and carbon pools in highland temperate forest landscape in Central Mexico. Forestry. 83, 497 -506.

Mikhailova, E.A., Post, C.J., 2006. Effects of land use on soil inorganic carbon stocks in the Russian Chernozem. Journal of Environmental Quality. 35, 1384-1388.

Milne, R., Brown, T.A., 1997. Carbon in the vegetation and soils of Great Britain. Journal of Environmental Management. 49, 413-433.

Ming, D.W., 2002. Carbonates, in: Lal, R. (Ed.) Encyclopedia of Soil Science. Marcel Dekker, New York, pp. 139-141.

Minkkinen, K., Laine, J., 1998. Long-term effect of forest drainage on the peat carbon stores of pine mires in Finland. Canadian Journal of Forest Research. 28, 1267-1275.

Minkkinen, K., Vasander, H., Jauhiainen, S., Karsisto, M., Laine, J., 1999. Post-drainage changes in vegetation composition and carbon balance in Lakkasuo mire, Central Finland. Plant and Soil. 207, 107-120.

Minkkinen, K., Byrne, K.A., Trettin, C.C., 2008. Climate impacts of peatland forestry. in: Strack, M. (Ed.), Peatlands and Climate Change. International Peat Society and Saarijärven Offset Oy, Saarijärvi, Finland. pp. 98-122.

Mishurov, M., Kiely, G., 2010. Nitrous oxide flux dynamics of grassland undergoing afforestation. Agriculture, Ecosystems & Environment. 139, 59-65.

Morris, S.J., Bohm, S., Haile-Mariam, S., Paul, E.A., 2007. Evaluation of carbon accrual in afforested agricultural soils. Global Change Biology. 13, 1145-1156.

Motavalli, P.P., Palm, C.A., Parton, W.J., Elliott, E.T., Frey, S.D., 1995. Soil pH and organic C dynamics in tropical forest soils: Evidence from laboratory and simulation studies. Soil Biology and Biochemistry. 27, 1589-1599.

Murty, D., Kirschbaum, M.U.F., McMurtrie, R.E., McGilvray, H., 2002. Does conversion of forest to agricultural land change soil carbon and nitrogen? A review of the literature. Global Change Biology. 8, 105–123.

Nelson, D.W., Sommers, L.E., 1996. Total carbon, organic carbon, and organic matter. in: Sparks,D.L., Page, A.L., Helmke, P.A., Loeppert, RH., Soltanpour, P.N., Tabatabai, M.A., Johnston, C.T., Sumner, M.ethods of Soil Analysis, Part 3 Chemical Methods. ASA and SSSA, Madison, WI, USA.

NFI, 2007a. National Forest Inventory - Republic of Ireland - Results. Forest Service, Department of Agriculture, Fisheries and Food, Johnstown Castle Estate, Co Wexford, Ireland.

NFI, 2007b. National Forest Inventory - Republic of Ireland - Methodology. Forest Service, Department of Agriculture, Fisheries and Food, Johnstown Castle Estate, Co Wexford, Ireland.

NFI, 2007c. National Forest Inventory - Republic of Ireland - Proceedings of NFI Conference. Forest Service, Department of Agriculture, Fisheries and Food, Johnstown Castle Estate, Co Wexford.

Nieuwenhuis, M., Tobin, B., Gardiner, P., Olajuyigbe, S., 2011. CARBIFOR II. Biomass Allocation-WWP1. http://www.ucd.ie/carbifor/index.html.

Nohrstedt, H.O., 2001. Response of coniferous forest ecosystems on mineral soils to nutrient additions: a review of Swedish experiences. Scandinavian Journal of Forest Research. 16, 555–573.

Nouvellon, Y., Epron, D., Kinana, A., Hamel, O., Mabiala, A., D'Annunzio, R., Deleporte, P., Saint-André, L., Marsden, C., Roupsard, O., Bouillet, J.-P., Laclau, J.-P., 2008. Soil CO_2 effluxes, soil carbon balance, and early tree growth following savannah afforestation in Congo: Comparison of two site preparation treatments. Forest Ecology and Management. 255, 1926–1936.

Örlander, G., Egnell, G., Albrektson, A., 1996. Long-term effects of site preparation on growth in Scots pine. Forest Ecology and Management. 86, 27–37.

Ostertag, R., Marín-Spiotta, E., Silver, W.L., Schulten, J., 2008. Litterfall and decomposition in relation to soil carbon pools along a secondary forest chronosequence in Puerto Rico. Ecosystems. 11, 701–714.

Pare, D., Bergeron, Y., 1995. Above-ground biomass accumulation along a 230-year chronosequence in the southern portion of the Canadian boreal forest. Journal of Ecology. 83, 1001–1007.

Parfitt, R.L., Percival, H.J., Dahlgren, R.A., Hill, L.F., 1997. Soil and solution chemistry under pasture and radiata pine in New Zealand. Plant and Soil. 191, 279-290.

Parker, J.L., Fernandez, I.J., Rustad, L.E., Norton, S.A., 2001. Effects of nitrogen enrichment, wildfire, and harvesting on forest-soil carbon and nitrogen. Soil Science Society of America Journal. 65, 1248-1255.

Paul, E.A., Morris, S.J., Six, J., Paustian, K., Gregorich, E.G., 2003. Interpretation of soil carbon and nitrogen dynamics in agricultural and afforested soils. Soil Science Society of America Journal. 67, 1620.

Paul, K.I., Polglase, P.J., Nyakuengama, J.G., Khanna, P.K., 2002. Change in soil carbon following afforestation. Forest Ecology and Management. 168, 241-257.

Peichl, M., Leava, N.A., Kiely, G., 2011a. Above- and belowground ecosystem biomass, carbon and nitrogen allocation in recently afforested grassland and adjacent intensively managed grassland. Unpublished Work.

Peichl, M., Leava, N.A., Kiely, G., 2011b. Initial changes in ecosystem production and respiration following afforestation of an intensively managed grassland. Unpublished Work.

Peng, Y., Thomas, S.C., Tian, D., 2008. Forest management and soil respiration: Implications for carbon sequestration. Environmental Reviews. 16, 93-111.

Peri, P.L., Gargaglione, V., Martínez Pastur, G., Lencinas, M.V., 2010. Carbon accumulation along a stand development sequence of *Nothofagus antarctica* forests across a gradient in site quality in Southern Patagonia. Forest Ecology and Management. 260, 229-237.

Perrott, K.W., Ghani, A., O'Connor, M.B., Waller, J.E., 1999. Tree stocking effects on soil chemical and microbial properties at the Tikitere Agroforestry Research Area. New Zealand Journal of Forestry Science. 29, 116-130.

Piccolo, M.C., Neill, C., Cerri, C.C., 1994. Natural abundance of 15 N in soils along forest-to-pasture chronosequences in the western Brazilian Amazon Basin. Oecologia. 99, 112–117.

Pilcher, J., R, Mac an tSaoir, S., 1995. Wood, Trees Forests in Ireland. Royal Irish Academy, Dublin, Ireland.

Pinno, B.D., Bélanger, N., 2008. Ecosystem carbon gains from afforestation in the boreal transition ecozone of Saskatchewan (Canada) are coupled with the devolution of black chernozems. Agriculture, Ecosystems & Environment. 123, 56–62.

Post, W.M., Kwon, K.C., 2000. Soil carbon sequestration and land-use change: processes and potential. Global Change Biology. 6, 317-327.

Post, W.M., Izaurralde, R.C.C., Mann, L.K., Bliss, N., 2001. Monitoring and verifying changes of organic carbon in soil. Climatic Change. 51, 73-99.

Ray, D., Xenakis, G., Semmler, T., Black, K., 2008. The impact climate change on forests in Ireland and some option for adaptation. in: Hendrick, E., Black, K., Forests, Carbon and Climate Change - Local and International Perspectives. COFORD, Dublin, Ireland. pp. 27-33.

Renou, F., Farrell, E.P., 2005. Reclaiming peatlands for forestry: the Irish experience, in: Stanturf, J.A. and Madsen, P.A. (Eds.). Restoration of Boreal and Temperate Forests. CRC Press, Boca Raton, pp. 541-557.

Resh, S.C., Binkley, D., Parrotta, J.A., 2002. Greater soil carbon sequestration under nitrogen-fixing trees compared with Eucalyptus species. Ecosystems. 5, 217–231.

Reynolds, B., 2007. Implications of changing from grazed or semi-natural vegetation to forestry for carbon stores and fluxes in upland organo-mineral soils in the UK. Hydrology and Earth System Sciences. 11, 61-76.

Richter, D.D., Markewitz, D., Trumbore, S.E., Wells, C.G., 1999. Rapid accumulation and turnover of soil carbon in a re-establishing forest. Nature. 400, 56–58.

Ritter, E., 2007. Carbon, nitrogen and phosphorus in volcanic soils following afforestation with native birch (*Betula pubescens*) and introduced larch (*Larix sibirica*) in Iceland. Plant and Soil. 295, 239–251.

Romanyà, J., Cortina, J., Falloon, P., Coleman, K., Smith, P., 2000. Modelling changes in soil organic matter after planting fast-growing *Pinus radiata* on Mediterranean agricultural soils. European Journal of Soil Science. 51, 627-641.

Rosa, E., Larocque, M., Pellerin, S., Gagné, S., Fournier, B., 2009. Determining the number of manual measurements required to improve peat thickness estimations by ground penetrating radar. Earth Surface Processes and Landforms. 34, 377-383.

Ross, D.J., Tate, K.R., Scott, N.A., Feltham, C.W., 1999. Land-use change: effects on soil carbon, nitrogen and phosphorus pools and fluxes in three adjacent ecosystems. Soil Biology and Biochemistry. 31, 803–813.

Saby, N., Bellamy, P.H., Morvan, X., Arrouays, D., Jones, R.J.A., Verheijen, F.G.A., Kibblewhite, M.G., Verdoodt, A., Üveges, J.B., Freudenschuß, A., Simota, C., 2008. Will European soil-monitoring networks be able to detect changes in topsoil organic carbon content? Global Change Biology. 14, 2432–2442.

Saiz, G., Byrne, K.A., Butterbach-Bahl, K., Kiese, R., Blujdea, V., Farrell, E.P., 2006. Stand age-related effects on soil respiration in a first rotation Sitka spruce chronosequence in central Ireland. Global Change Biology. 12, 1007–1020.

Sakovets, V.V., Germanova, N.I., 1992. Changes in the carbon balance of forested mires due to drainage. Suo. 43, 249-252.

Sartori, F., Lal, R., Ebinger, M.H., Eaton, J.A., 2007. Changes in soil carbon and nutrient pools along a chronosequence of poplar plantations in the Columbia Plateau, Oregon, USA. Agriculture, Ecosystems & Environment. 122, 325-339.

Schlesinger, W.H., 2002. Inorganic carbon and the global carbon cycle. in: Lal, R. (Ed.) Encyclopedia of Soil Science. Marcel Dekker, New York, pp. 706-708.

Schmidt, M.G., Macdonald, S.E., Rothwell, R.L., 1996. Impacts of harvesting and mechanical site preparation on soil chemical properties of mixed-wood boreal forest sites in Alberta. Canadian Journal of Soil Science. 76, 531–540.

Scott, N.A., Tate, K.R., Ross, D.J., Parshotam, A., 2006. Processes influencing soil carbon storage following afforestation of pasture with *Pinus radiata* at different stocking densities in New Zealand. Australian Journal of Soil Research. 44, 85-96.

Shi, J., Cui, L., 2010. Soil carbon change and its affecting factors following afforestation in China. Landscape and Urban Planning. 98, 75-85.

Shi, W., Yao, H., Bowman, D., 2006. Soil microbial biomass, activity and nitrogen transformations in a turfgrass chronosequence. Soil Biology and Biochemistry. 38, 311–319.

Shotbolt, L., Anderson, A.R., Townend, J., 1998. Changes to blanket bog adjoining forest plots at Bad a' Cheo, Rumster Forest, Caithness. Forestry. 71, 311 -324.

Six, J., Callewaert, P., Lenders, S., De Gryze, S., Morris, S.J., Gregorich, E.G., Paul, E.A., Paustian, K., 2002a. Measuring and understanding carbon storage in afforested soils by physical fractionation. Soil Science Society of America Journal. 66, 1981-1987.

Six, J., Conant, R.T., Paul, E.A., Paustian, K., 2002b. Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. Plant and Soil. 241, 155–176.

Smal, H., Olszewska, M., 2008. The effect of afforestation with Scots pine (*Pinus silvestris* L.) of sandy post-arable soils on their selected properties. II. Reaction, carbon, nitrogen and phosphorus. Plant and Soil. 305, 171-187.

Smith, P., Falloon, P.D., Körschens, M., Shevtsova, L.K., Franko, U., Romanenkov, V., Coleman, K., Rodionova, V., Smith, J.U., Schramm, G., 2002. EuroSOMNET–a European database of long-term experiments on soil organic matter: the WWW metadatabase. The Journal of Agricultural Science. 138, 123–134.

Smith, P., 2004. How long before a change in soil organic carbon can be detected? Global Change Biology. 10, 1878–1883.

Sollins, P., Homann, P., Caldwell, B.A., 1996. Stabilization and destabilization of soil organic matter: mechanisms and controls. Geoderma. 74, 65–105.

Sottocornola, M., 2007. Four Years of Observations of Carbon Dioxide Fluxes, Water and Energy Budgets, and Vegetation Patterns in an Irish Atlantic Blanket Bog. Ph.D. Thesis, Department of Civil and Environmental Engineering, University College Cork, Cork, Ireland, http://www.ucc.ie/hydromet/files/Sottocornola.2007.pdf.

Sottocornola, M., Kiely, G., 2010. Hydro-meteorological controls on the CO_2 exchange variation in an Irish blanket bog. Agricultural and Forest Meteorology. 150, 287-297.

Strong, W.L., La Roi, G.H., 1983. Root-system morphology of common boreal forest trees in Alberta, Canada. Canadian Journal of Forest Research. 13, 1164–1173.

Tang, J., Bolstad, P.V., Martin, J.G., 2009. Soil carbon fluxes and stocks in a Great Lakes forest chronosequence. Global Change Biology. 15, 145-155.

Taylor, A.R., Wang, J.R., Chen, H.Y.H., 2007. Carbon storage in a chronosequence of red spruce (*Picea rubens*) forests in central Nova Scotia, Canada. Canadian Journal of Forest Research. 37, 2260-2269.

Thuille, A., Buchmann, N., Schulze, E.-D., 2000. Carbon stocks and soil respiration rates during deforestation, grassland use and subsequent Norway spruce afforestation in the Southern Alps, Italy. Tree Physiology. 20, 849 -857.

Tobin, B., Black, K., Osborne, B., Reidy, B., Bolger, T., Nieuwenhuis, M., 2006. Assessment of allometric algorithms for estimating leaf biomass, leaf area index and litter fall in different-aged Sitka spruce forests. Forestry. 79, 453.

Tomlinson, R.W., Davidson, L., 2000. Estimates of carbon stores in four Northern Irish lowland raised bogs. Suo. 51, 169-179.

Tomlinson, R.W., 2005. Soil carbon stocks and changes in the Republic of Ireland. Journal of Environmental Management. 76, 77-93.

Tomlinson, R.W., 2006. CLIMATE CHANGE – Land Use, Land-Use Change and Carbon Stocks. Synthesis Report. Environmental Protection Agency, Ireland.

Torn, M.S., Trumbore, S.E., Chadwick, O.A., Vitousek, P.M., Hendricks, D.M., 1997. Mineral control of soil organic carbon storage and turnover. Nature. 389, 170–173.

Tremblay, S., Périé, C., Ouimet, R., 2006. Changes in organic carbon storage in a 50 year white spruce plantation chronosequence established on fallow land in Quebec. Canadian Journal of Forest Research. 36, 2713-2723.

Trumbore, S.E., 1997. Potential responses of soil organic carbon to global environmental change. Proceedings of the National Academy of Sciences of the United States of America. 94, 8284.

Turner, J., Lambert, M., 2000. Change in organic carbon in forest plantation soils in eastern Australia. Forest Ecology and Management. 133, 231-247.

Turner, J., Lambert, M.J., Johnson, D.W., 2005. Experience with patterns of change in soil carbon resulting from forest plantation establishment in eastern Australia. Forest Ecology and Management. 220, 259–269.

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

UNFCCC, 1992. United Nations Framework Convention on Climate Change. Palais des Nations, Geneva. http://www.unfccc.de/index.html.

Uselman, S.M., Qualls, R.G., Lilienfein, J., 2007. Fine root production across a primary successional ecosystem chronosequence at Mt. Shasta, California. Ecosystems. 10, 703–717.

Uselman, S.M., Qualls, R.G., Lilienfein, J., 2009. Production of total potentially soluble organic C, N, and P across an ecosystem chronosequence: root versus leaf litter. Ecosystems. 12, 240–260.

van Bergen, P.F., Nott, C.J., Bull, I.D., Poulton, P.R., Evershed, R.P., 1998. Organic geochemical studies of soils from the Rothamsted Classical Experiments--IV. Preliminary results from a study of the effect of soil pH on organic matter decay. Organic Geochemistry. 29, 1779-1795.

VandenBygaart, A.J., 2006. Monitoring soil organic carbon stock changes in agricultural landscapes: Issues and a proposed approach. Canadian Journal of Soil Science. 86, 451-463.

VandenBygaart, A.J., Angers, D.A., 2006. Towards accurate measurements of soil organic carbon stock change in agroecosystems. Canadian Journal of Soil Science. 86, 465-471.

Vasander, H., Kettunen, A., 2006. Carbon in boreal peatlands, in: Wieder, R.K. & Vitt, D.H. (Eds) Boreal Peatland Ecosystems. Ecological Studies 188, Springer-Verlag, Heidelberg, Germany. pp. 165-194.

Veldkamp, E., Davidson, E., Erickson, H., Keller, M., Weitz, A., 1999. Soil nitrogen cycling and nitrogen oxide emissions along a pasture chronosequence in the humid tropics of Costa Rica. Soil Biology and Biochemistry. 31, 387–394.

Vesterdal, L., Raulund-Rasmussen, K., 1998. Forest floor chemistry under seven tree species along a soil fertility gradient. Canadian Journal of Forest Research. 28, 1636–1647.

Vesterdal, L., Ritter, E., Gundersen, P., 2002. Change in soil organic carbon following afforestation of former arable land. Forest Ecology and Management. 169, 137-147.

Vesterdal, L., Schmidt, I.K., Callesen, I., Nilsson, L.O., Gundersen, P., 2008. Carbon and nitrogen in forest floor and mineral soil under six common European tree species. Forest Ecology and Management. 255, 35–48.

von Lutzow, M., Kogel-Knabner, I., Ekschmitt, K., Matzner, E., Guggenberger, G., Marschner, B., Flessa, H., 2006. Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil conditions – a review. European Journal of Soil Science. 57, 426–445.

von Post, L., 1924. Das genetische System der organogenen Bildungen Schwedens. In: Comité International de Pédologie, IVème commission (commission pour la nomenclature et la classification des sols, commission pour l'Europe, président: B. Frosterus) (Ed.).

Mémoires sur la Nomenclature et la Classification des Sols, Helsingfors/Helsinki. pp. 287-304.

Wang, C.M., Ouyang, H., Shao, B., Tian, Y.Q., Zhao, J.G., Xu, H.Y., 2006. Soil carbon changes following afforestation with Olga Bay larch (*Larix olgensis* Henry) in northeastern China. Journal of Integrative Plant Biology. 48, 503–512.

Weiss, D., Shotyk, W., Rieley, J., Page, S., Gloor, M., Reese, S., Martinez-Cortizas, A., 2002. The geochemistry of major and selected trace elements in a forested peat bog, Kalimantan, SE Asia, and its implications for past atmospheric dust deposition. Geochimica et Cosmochimica Acta. 66, 2307-2323.

Wellock, M.L., LaPerle, C.M., Peichl, M., Kiely, G., 2011a. Changes in ecosystem carbon stocks in a grassland-ash (*Fraxinus excelsior* L.) afforestation chronosequence in Ireland. Unpublished Work.

Wellock, M.L., Reidy, B., LaPerle, C.M., Bolger, T., Kiely, G., 2011b. Soil carbon stocks of afforested peat soils in Ireland. Unpublished Work.

White, L.L., Zak, D.R., Barnes, B.V., 2004. Biomass accumulation and soil nitrogen availability in an 87-year-old *Populus grandidentata* chronosequence. Forest Ecology and Management. 191, 121–127.

Wuest, S.B., 2009. Correction of bulk density and sampling method biases using soil mass per unit area. Soil Science Society of America Journal. 73, 312-316.

Xu, X., Wen, L., Zhang, C., Kiely, G., 2011. Estimation of Soil Organic Carbon (SOC) stock and its spatial distribution in the Republic of Ireland. Unpublished Work.

Yanai, R.D., Currie, W.S., Goodale, C.L., 2003. Soil carbon dynamics after forest harvest: an ecosystem paradigm reconsidered. Ecosystems. 6, 197-212.

Yang, X.-M., Wander, M.M., 1999. Tillage effects on soil organic carbon distribution and storage in a silt loam soil in Illinois. Soil and Tillage Research. 52, 1-9.

Zavaleta, E.S., Kettley, L.S., 2006. Ecosystem change along a woody invasion chronosequence in a California grassland. Journal of Arid Environments. 66, 290-306.

Zerva, A., Ball, T., Smith, K.A., Mencuccini, M., 2005. Soil carbon dynamics in a Sitka spruce (*Picea sitchensis* (Bong.) Carr.) chronosequence on a peaty gley. Forest Ecology and Management. 205, 227-240.

Zerva, A., Mencuccini, M., 2005. Carbon stock changes in a peaty gley soil profile after afforestation with Sitka spruce (*Picea sitchensis*). Annals of Forest Science. 62, 873-880.

Zinn, Y.L., Resck, D.V.S., da Silva, J.E., 2002. Soil organic carbon as affected by afforestation with Eucalyptus and Pinus in the Cerrado region of Brazil. Forest Ecology and Management. 166, 285–294.

Zou, X., Bashkin, M., 1998. Soil carbon accretion and earthworm recovery following revegetation in abandoned sugarcane fields. Soil Biology and Biochemistry. 30, 825–830.