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Carbon storage in Orchards

Anthony, Rosemary

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Carbon Storage in Orchards



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A thesis submitted to Bangor University by Rosemary Anthony in
candidature for the degree of Doctor of Philosophy

School of Environment, Natural Resources and Geography
Bangor University
Bangor
Gwynedd
LL57 2UW

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Abstract

Current concern over climate change requires that carbon (C) sources and sinks within managed landscapes are quantified so that they can be actively managed to help reduce net greenhouse gas (GHG) emissions. While some data exist for C storage in grassland and forests, potential for C sequestration in orchards is largely unknown. Initial C inventory studies in apple (*Malus*) orchards determined that C sequestration in soil and trees increased with stand age. This result conflicts with current commercial management which requires trees to be replaced every 30 years to reduce disease risk. The grubbing-out process, where complete orchard trees are removed, liberated approximately 0.01 t C ha⁻¹ from the soil, which is very small in comparison to the C loss from removal of biomass (up to 25 t C ha⁻¹ above-ground, ca. 5 t C ha⁻¹ below-ground) and that held in the soil (ca. 80 t C ha⁻¹). The incorporation of biochar produced from the grubbed-up apple trees offers orchards the opportunity to enhance C storage. However long-term impacts of biochar amendments on the environment remain unknown. An experiment was established to monitor Bramley apple trees in varying levels of biochar amendment (0, 5, 25 and 50 t ha⁻¹). The results suggested that biochar had no negative effects on tree productivity and soil quality. Investigations into density, surface area, pore size distribution and water sorption were carried out to characterise the *Malus*-derived char. Orchard carbon footprints have been carried out to determine orchard GHG emissions and identify potential areas of C savings. Findings suggest that footprints of UK apple production are comparatively low, ranging from 0.037 to 0.182 kg CO₂e per kg produce. In conclusion, there is potential within UK orchards to sequester C; full C budgets need to be carried out to determine annual C sequestration.

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Abbreviations

AF	Airborne fraction
ANOVA	Analysis of variance
BAP	Biodiversity Action Plan
BBKA	British Beekeeper's Association
BET	Brunauer-Emmett-Teller method
BIGFA	British Independent Fruit Growers Association
BJH	Barett, Joyner and Halenda method
BSI	British Standards Institute
C	Carbon
Ca	Calcium
CCP	Climate change programme
CEC	Cation exchange capacity
CH ₄	Methane
CO ₂	Carbon dioxide
CO ₂ e	Carbon dioxide equivalents
dbh	Diameter at breast height
Defra	Department for Environment, Food and Rural Affairs
DOC	Dissolved organic carbon
DON	Dissolved organic nitrogen
DVS	Dynamic vapour sorption
EC	Electrical conductivity
EF	Emission factor
EMC	Equilibrium moisture content
ERM	Extra-radical mycelium
FACE	Free-Air Carbon dioxide Enrichment
FPC	Fresh Produce Consortium
FSA	Food Standards Agency
GHG(s)	Greenhouse gas(es)
GPP	Gross primary production
GWP	Global warming potential
H	Hydrogen
HDC	Horticultural Development Company
HFC(s)	Hydrofluorocarbon(s)
Hg	Mercury
HH	Hailwoo-Horrobin model
HPM	Home production market
IBI	International Biochar Initiative
IPCC	Intergovernmental Panel on Climate Change
K	Potassium
K ₂ O	Potassium oxide
LAI	Leaf Area Index
LCA	Life Cycle Assessment
LEA	Local Education Authority
LOI	Loss-on-ignition

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LUC	Land-use change
MAFF	Ministry of Agriculture, Fisheries and Food
MC	Moisture content
N	Nitrogen
N ₂ O	Nitrous oxide
Na	Sodium
NACM	National Association of Cider Makers
NCE	Net carbon exchange
NPP	Net primary production
O ₂	Oxygen
OH	Hydroxyl
P	Phosphorus
P ₂ O ₅	Superphosphate
PAHs	Polycyclic aromatic hydrocarbons
PAS 2050	Publically Available Standard 2050
PFC(s)	Perfluorocarbon(s)
R & D	Research and Development
R _A	Autotrophic respiration
R _H	Heterotrophic respiration
R _S	Soil respiration
RH	Relative humidity
S	Sulphur
SF ₆	Sulphur hexafluoride
SOC	Soil organic carbon
SOM	Soil organic matter
TEV	Total economic value
UN	United Nations
WAPA	World Apple and Pear Association

CHAPTER ONE

INTRODUCTION

1.1 The changing global climate

There is a growing public and political concern regarding the impact of human activity on climate change and the loss of biodiversity it has and will continue to cause (Gitay *et al.*, 2002). Climate change is causing rapid changes that are accelerating at a worrying rate due to human interaction, including land use change, an increase in industrial activity causing air, soil and water pollution, habitat fragmentation, species exploitation, introduction of alien species, increased production of greenhouse gases (GHGs) due to the combustion of fossil fuels and ozone depletion (Gielen and Ceulemans, 2001; Gitay *et al.*, 2002). The global carbon (C) balance has been disturbed due to major increases in anthropogenic activity since the Industrial Revolution, which have caused a rise in C emissions and an increase in the levels of atmospheric carbon dioxide (CO₂) and other GHGs, such as methane (CH₄) and nitrous oxide (N₂O). A series of numerous other effects have also been observed, which vary worldwide including changing temperatures, rising sea-levels, nitrogen deposition, increased frequency and intensity of storms, changed seasonal habits and rainfall variation (Cannell *et al.*, 1998; Gielen and Ceulemans, 2001; Gitay *et al.*, 2002). The global climate is moving towards a trend in warming, but, this is not the only impact of greenhouse gases on terrestrial ecosystems. Other effects are likely to occur and may change the ecological balance of ecosystems. They may include an increased length of the growing season, increased plant growth rates leading to the modification of ecosystem composition due to the competitive balance between species being disturbed, ecosystem function, species diversity, increased precipitation, increased temperature, and decreased cloud cover (Canadell *et al.*, 2007b; Malhi *et al.*, 2003). CO₂ is necessary for photosynthesis and increased levels are likely to lead to CO₂ fertilisation and increased net primary production (Körner *et al.*, 2007). Although the effects of the changes are unknown, it is thought that they are likely to have a cascade effect on species and ecosystems resulting in changes in phenology, reproductive and migration patterns, species distributions, length of growing seasons

and a change in the frequency and type of pest outbreaks, which may ultimately threaten biodiversity (Gitay *et al.*, 2002; Thuiller *et al.*, 2005).

Global climate change is due to the rapid growth in the use of fossil fuels, industrial processes and land-use change, which have produced CO₂ emissions at an accelerating rate, with their growth rate increasing from 1.1% yr⁻¹ for 1990-1999 to approximately 3.3% yr⁻¹ for 2000-2006 (Canadell *et al.*, 2007a; Malhi *et al.*, 2003; Raupach *et al.*, 2007). Global mean temperature has increased by 0.74°C±0.2°C between 1906 and 2005, with the Intergovernmental Panel on Climate Change (IPCC) predicting it to rise further over the next two decades at a rate of approximately 0.2°C per decade (IPCC, 2007a; Malhi *et al.*, 2003).

The rise in temperature, along with increased CO₂ deposition of approximately 1% annually throughout the globe and increased nitrogen (N) deposition will lead to a fertilising effect with increased rates of photosynthesis. This will result in an increase in productivity and yield of individual tree species and stand dynamics, whilst also having a positive effect on water use, wind resistance and timber quality (Broadmeadow, 2002; Broadmeadow *et al.*, 2005; Cannell *et al.*, 1998; Ceulemans *et al.*, 1999; Gitay *et al.*, 2002; Scarascia-Mugnozza *et al.*, 2001). It has been shown using Free-Air Carbon dioxide Enrichment (FACE) studies that there is a marked increase in primary production and that there is an increase in growth (leaf area and photosynthetic rate) under elevated CO₂ levels (Calfapietra *et al.*, 2003; Ceulemans *et al.*, 1999; Hoosbeek *et al.*, 2006).

The increase in temperature caused by global warming will bring early flushing, which then leads to risk of bud damage by unseasonal frosts, and summer drought can occur thereby incurring losses of newly established trees (Broadmeadow, 2002; Gitay *et al.*, 2002). There is a positive correlation between temperature and soil respiration, which releases CO₂ from soils (Lloyd and Taylor, 1994).

Winters in the UK are expected to have a higher incidence of rainfall, with increases in the range of up to 10 and 30% over the majority of the country, despite summer droughts. This is expected to lead to an increase in soil moisture and higher incidence of

water logging, which will affect tree stability and may affect root survival (Broadmeadow, 2002). Regional changes in climate and growing conditions are predicted and effects are already being seen in some parts of the world. This may have a profound effect on species composition and cause habitat fragmentation by forcing many species to migrate northwards. Some may not survive if adaptation does not occur quickly enough or the genetic capacity is not present (Broadmeadow, 2002; Broadmeadow *et al.*, 2005; Dresner *et al.*, 2006; Gitay *et al.*, 2002; Thuiller *et al.*, 2005).

Fossil fuel combustion is responsible for approximately 75% of the anthropogenic C emissions and is particularly significant as it results in removal of C from the biosphere and long-term C stores (Malhi *et al.*, 2003). Mean global atmospheric CO₂ levels have increased at a progressively faster rate each year from 280 ppm in the 1700s to the current figure of > 400 ppm, with a recent rapid growth in atmospheric CO₂ (> 2 ppm yr⁻¹; Canadell *et al.*, 2007a; Raupach *et al.*, 2007). The present concentration is the highest it has been during the last 650,000 years and is predicted to rise to between 490 and 1260 ppm by the year 2100 as concentrations continue to rise (Petit *et al.*, 1999; Pittock, 2005; Malhi *et al.*, 2003; Scarascia-Mugnozza *et al.*, 2001; Siegenthaler *et al.*, 2005). Approximately 63% of the gaseous radiative forcing (the warming effect on the earth caused by GHGs absorbing and trapping some of the long-wave thermal radiation in our atmosphere instead of emitting it out towards space) responsible for anthropogenic climate change is caused by atmospheric CO₂ therefore reducing the emissions or finding methods of mitigation are essential (Malhi *et al.*, 2003; Raupach *et al.*, 2007). Under the Kyoto Protocol, an international agreement for industrialised countries to cut their GHG emissions, the European Union is committed to reducing CO₂ emissions by 8% compared to the 1990 baseline levels by 2012 and within this agreement, the UK is committed to a 12.5% reduction in their emissions (Smith *et al.*, 2000).

1.1.1 The Kyoto Protocol

Most major international treaties addressing ecosystem protection fail to affect large-scale conservation due to the lack of meaningful binding obligations and the finance to implement strategies (Swingland, 2003). However, in 1992 the United Nations held a Framework Convention on Climate Change, which led to the signing of the Kyoto

Protocol, which was agreed in 1997 and came into force as a legally binding treaty on 16th February 2005 (a copy of the Protocol is available at: UNFCCC, 2009). The protocol is an international agreement that sets binding targets for 37 industrialized countries to reduce GHG emissions and provides economic incentives to achieve this (Swingland, 2003; UNFCCC, 2009). GHGs targeted include CO₂, CH₄, N₂O, hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulphur hexafluoride (SF₆). These gases, with CO₂ being the most important driver, have a significant role in global warming as they trap solar heat, which has increased global temperature and warmed up the global climate (Malhi *et al.*, 2003; UNFCCC, 2009). The Kyoto objective is to achieve a lowering of GHG concentrations to a level of stabilisation in the atmosphere, which would prevent any further anthropogenic damage to the climate system (UNFCCC, 2009). Targets of the treaty involve cutting C emissions and/or engaging in emissions trading. Industrialised countries have committed to cutting emissions to 5% below the 1990 baseline levels by 2008-2012 (UNFCCC, 2009). Figures from the UN predict that not all the countries signed up are on course to meet their targets and after little progress has been made by some, it is thought that collective emissions may well be 10% above the 1990 levels by 2010 rather than below them (UNFCCC, 2009).

1.1.2 Carbon sinks and sources

There are four main C stores on earth (Figure 1.1): geological, oceanic, terrestrial and atmospheric reserves, with the atmosphere being the smallest yet highly significant component as any change in the level of C in this store is directly linked with changes in climate (Malhi *et al.*, 2003). Less than half of the total cumulative anthropogenic CO₂ emissions remains in the atmosphere, the rest are taken up by terrestrial and ocean sinks (Le Quéré *et al.*, 2007; Raupach *et al.*, 2007). This C sequestration from terrestrial and aquatic systems demonstrates the importance of ecosystem services provided by forests, agricultural lands, and the ocean (Swingland, 2003). They not only provide important habitats but can provide a practical and available method of mitigating C from the atmosphere, so conservation and protection of these ecosystems must be encouraged (Swingland, 2003). Natural land and ocean sinks removing part of the anthropogenic CO₂ results in the annual atmospheric CO₂ increment being substantially smaller than that of increment in anthropogenic emissions (Canadell *et al.*, 2007a). The

terrestrial biosphere is an important component in slowing the build-up of atmospheric CO₂ as it removes approximately one third of CO₂ emitted from the atmosphere (Canadell *et al.*, 2007b; Canadell and Raupach, 2008; Hurteau *et al.*, 2009).

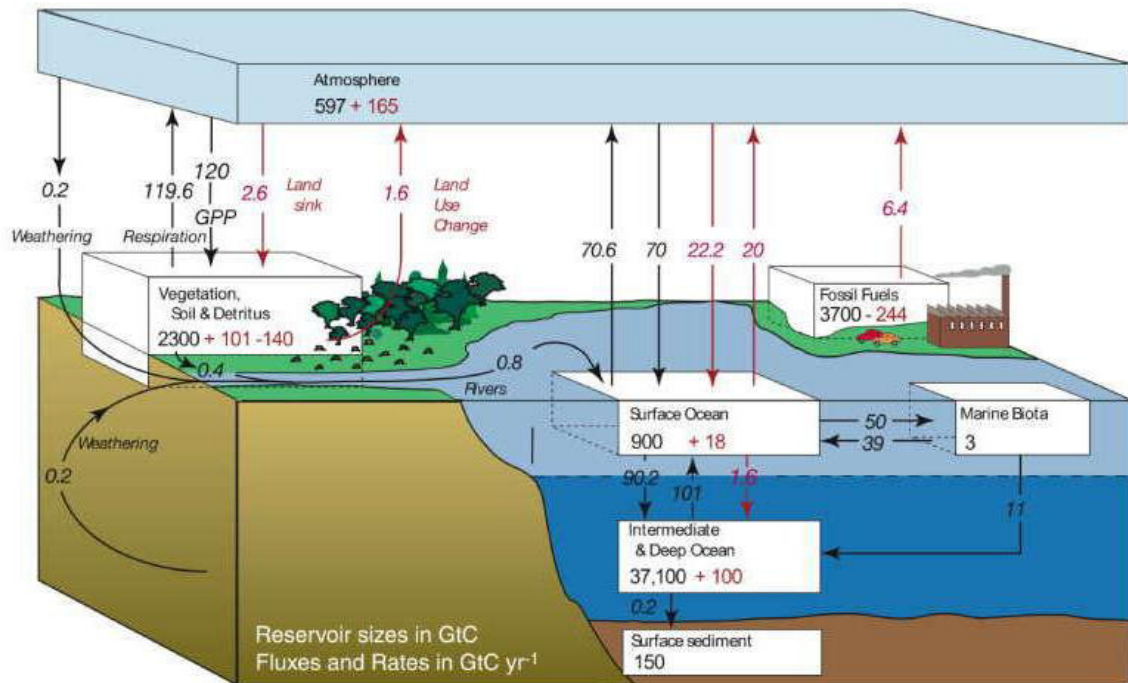


Figure 1.1. The global C cycle for the 1990s, showing the main annual fluxes in GtC yr⁻¹: pre-industrial 'natural' fluxes in black and 'anthropogenic' fluxes in red (taken from IPCC, 2007c).

An important ecosystem in terms of mitigation is that of forests, which store large reservoirs of C above and below ground, together holding double the amount of the C in the atmosphere (Brown *et al.*, 2003; Canadell and Raupach, 2008; Hurteau *et al.*, 2009). The C stocks accumulate in various above- and below-ground pools (foliage, branches, standing timber, roots, necromass, litter, woody debris, soil organic matter (SOM) and forest products) within the ecosystem through the absorption of atmospheric CO₂ and its assimilation into biomass (Malhi *et al.*, 2003). Litter decomposition is a vital process in ecosystem functioning and has important implications on C cycling (Gillon *et al.*, 1999). The long-term rise in atmospheric C will have an effect on forests and trees as they are instrumental in C sequestration (Godbold *et al.*, 2006). If there are no limiting resources such as N, the increased photosynthesis, as a result of increased CO₂ levels, may lead to an increase in the C storage capacity in the wood as a direct result of increased biomass of forests due to a C limitation under current levels of CO₂ for trees that use the C₃ photosynthesis mechanism (DeLucia *et al.*, 1999). The vegetation in the ecosystem

absorbs C from the atmosphere and between 30-50% of this is released back into the atmosphere as a by-product of respiration, but the remaining C is fixed as organic matter above- or below-ground (net primary photosynthesis) (Malhi *et al.*, 2003). This C sequestration service is thought to play a role in the mitigation of the rising levels of atmospheric CO₂ and can potentially reduce the anthropogenic increase in atmospheric CO₂. Therefore it is recognised in GHG protocols and there is interest from cap-and-trade mechanisms due to the potential to manage the C sink of the forested systems (Del Galdo *et al.*, 2003; DeLucia *et al.*, 1999; Hurteau *et al.*, 2009; Schlesinger and Lichter, 2001; Swingland, 2003). Forest gross photosynthesis, which accounts for 50% of total terrestrial photosynthesis (approximately 60 Pg C yr⁻¹), is responsible for annually cycling approximately half of the atmospheric CO₂ stock (Swingland, 2003). This C cycling within forests had been maintained at a constant level since the end of the last ice age but underwent severe disruption with the onset of the Industrial Revolution and the subsequent addition of 480 Pg C into the atmosphere due to increased combustion of fossil fuels and land-use change (Swingland, 2003). Forestry activities that are shown to reduce C emissions, along with land-use and land-management change, can be included under the Kyoto targets, as the protocol aims to offset emissions by the demonstrable removal of C from the atmosphere (Brown *et al.*, 2003; Smith *et al.*, 2000). The regrowth of forests on abandoned lands and from CO₂ fertilisation resulting from increased levels in the atmosphere, are evidence for natural vegetation acting as sinks and absorbing C. This vegetation biosphere sink was estimated to sequester 3.2 Pg C yr⁻¹ in the 1990s and will affect the dynamics, structure and biodiversity of the forest ecosystem (Swingland, 2003). Forests can switch to becoming a source of atmospheric C during times of disturbance, either due to human or natural causes, such as the use of poor harvesting practices or wildfires, respectively (Brown *et al.*, 2003). There are four major strategies involving forests and their activities to mitigate C emissions and slow the rise of atmospheric CO₂ levels. These are:

- i. Increase the area of forest cover by afforestation or reforestation
- ii. Increase the C density of existing forests (longer harvesting cycles or reduced disturbance) until equilibrium is reached
- iii. Expand the use of products that sustainably replace fossil-fuel CO₂ emissions

- iv. Reduce emissions from deforestation and land-use change (Canadell and Raupach, 2008; Hurteau *et al.*, 2009; Smith *et al.*, 2000).

The balance between anthropogenic C emissions and the dynamics of terrestrial and ocean processes is reflected in the rising levels of atmospheric CO₂ and will determine the mitigation requirements necessary to stabilise the situation. The change and variability in climate has led to increased changes in land and ocean fluxes. While sinks play a vital role it is thought there may be a decline in the efficiency of CO₂ sinks on land and oceans in absorbing anthropogenic emissions as there is evidence ($p > 0.05$) for a long-term (50 year) increase in the airborne fraction (AF) of CO₂ emissions (Canadell *et al.*, 2007a). The annual AF measures the relative efficiency of those sinks by using the ratio of the atmospheric CO₂ increase in a given year to that year's total emissions ($F_{FOSS} + F_{LUC}$) and is a function of the biological and physical processes that govern the CO₂ balance between terrestrial, ocean and atmospheric systems (Canadell *et al.*, 2007a). Although climate-C models predict an increasing AF, the magnitude being observed is far greater than the estimated value, which implies that C emissions have grown faster than CO₂ sinks (Canadell *et al.*, 2007a). There are three possible scenarios that could be causing this trend: the source may have intensified, the sink may have weakened or the sink may have transitioned to a source. The rising C emissions and reduction in sink strength characterize a C cycle that is generating a much stronger and faster climate forcing (Canadell *et al.*, 2007a).

The use of sinks to sequester C is only a component in stabilising atmospheric CO₂ concentrations (Swingland, 2003). There is no single land-management change solution to the mitigation of C emissions in the UK. However, an integrated approach combining land-management options and changes in energy supply, use and technology to increase the reduction in CO₂ emissions could potentially have a positive effect on climate change (Smith *et al.*, 2000; Swingland, 2003).

1.1.2.1 The role of perennial plantations in carbon sequestration

It has been suggested that forest plantations could play a key role in the mitigation of future climate change by being used as a measure to store C (Liao *et al.*, 2010). Potential to sequester C lies in the soil and the above- and belowground biomass (Metz *et al.*,

2007). Although anthropogenic woody plantations may have a lower biodiversity than natural forests, they will sequester CO₂ from the atmosphere into plant biomass and the soil (IPCC, 2001b). According to the IPCC, there is approximately 700 Mha of forestland and 345 Mha of plantations and agroforestry worldwide (IPCC, 2001b). It is suggested that land has the potential to sequester up to 319 Gt C by 2050 in global forests, while the mitigation potential of plantations could reach a maximum rate of 2.2 Gt C yr⁻¹ (IPCC, 2001b; Metz *et al.*, 2007). This is significant as the global atmospheric CO₂ increased by an average of 3.3±0.2 GtC yr⁻¹ during the 1990s (IPCC, 2001b). While plantations offer a potential store of C, uncertainties remain as storage is typically lower, by approximately 28% for total ecosystem C stock, than natural forest (Liao *et al.*, 2010). The rate of storage will depend on a number of different variables, including plantation variety, and variation is caused by climate, geography, species and stand age (Dewar and Cannell, 1992; Liao *et al.*, 2010). Despite there being uncertainty over the amount of C stored within the soil of plantations (due to the variation regarding factors such as variety and geographical area), it is accepted that afforestation may lead to increase C storage in the vegetation, thereby aiding the mitigation of climate change (Scott *et al.*, 1999).

1.2 Orchards, climate and carbon

Many orchards in the UK are being grubbed out in order to make way for new land uses which appear to have higher direct economic value associated with them, such as grazing, growing cereals and soft fruit and as a pressure for the need for new housing. However, are other factors that give orchards value and instead of just looking at the basic economic value of orchards, their total economic value (TEV) should be measured (Taplin, 2008). The TEV is the sum of the direct use values (consumption of products and its attributes), indirect use values (ecosystem functioning and biodiversity value), option values (having the resource for the future), bequest values (protection of habitats) and the existence values. In other words, the economic, environmental and social components of orchards must be considered, such as the roles that they play within biodiversity, climate change, soil quality, profit, local economic impact, tourism and social impact (Hereford Orchards, 2009; Taplin, 2008). It is difficult to understand the effects of climate and which factors are critical in apple orchards as their productivity varies greatly between different climates and even within the same climate from year to

year, however, any changes in the climate will have an effect on orchards and their productivity (Lakso *et al.*, 2000). This has been observed during recent years in the UK with unpredictable weather leading to poor apple quality and lower yield.

Malhi *et al.* (2003) state that “any activity that affects biomass amount in vegetation and soil has potential to sequester C from, or release C into, the atmosphere”. This applies to orchards, where the apple trees need to store C and other nutrients to support initial spring growth, and the system can contribute to climate change mitigation by the impact they have on global atmospheric CO₂ (Hereford Orchards, 2009; Lakso *et al.*, 1999). The orchard system removes CO₂ from the atmosphere, while C is locked-up in the growing tree biomass and accumulates in the organic matter of the soil. However, CO₂ is also emitted through the production side of the industry by burning fossil fuels to run the machinery to maintain, harvest, store and transport the fruit (Hereford Orchards, 2009).

The National Association of Cider Makers (NACM) carried out a study, published in 2008, on the carbon footprint of the UK cider industry. It aimed to quickly and cheaply establish an approximate carbon footprint for the whole industry in order to set a benchmark for reduction targets. It was found that there is a positive linear relationship between CO₂ and the volume of cider produced, thus enabling the prediction of the amount of emissions associated with any given volume of cider. The industry annual footprint is estimated at approximately 68,500 tonnes CO₂ (NACM, 2008).

The change in climate will have a large effect on the fruit production industry as the growth and yield is hugely dependent upon weather conditions throughout the year, any change will have an impact on the current year’s crop but also on the following year’s yield (Copas *et al.*, 2008). The unpredictability of the weather can impact at any stage throughout the growing process by causing failure of the fruit to set, poor juice quality, low yields and inadequate flower bud production thus affecting the following year’s crop (Copas *et al.*, 2008). Temperature, a key factor in fruit and shoot growth and a main environmental factor affecting the net C exchange (NCE), C balance and C partitioning in apple trees, is set to change with the changing climate (Calderón-Zavala *et al.*, 2004).

Key issues of concern for UK apple orchards due to climate change include a decrease in number of very cold winter days, leading to inadequate winter chill for dormancy and vernalization and an increase in frost incidence that will damage and affect crops (Copas *et al.*, 2008; Farming Futures, 2008). There are also concerns due to the arrival of new pests and diseases, which are more likely to survive due to a lack of winter “kill off”, pollination concerns due to the decline in the honey bee population and the potential loss of low-lying land due to increased flooding and rising sea levels (Beckenham, 2009; Copas *et al.*, 2008; Farming Futures, 2008). Fire blight (*Erwinia amylovora*) may become more widespread in warm, wet spring time, while the hotter, drier summers may encourage the occurrence of mildew (*Podosphaera leucotrica*) but may limit the development of scab (*Venturia inaequalis*), which may increase during the warmer, wetter autumn weather expected with climate change (Farming Futures, 2008). Extreme weather events, such as hailstorms, may influence the developing fruit in the orchard system.

There may be stop-start springs leading to weak blossom and a poor fruit set, coupled with unpredictable summer weather affecting cropping patterns and pollinator activity (Farming Futures, 2008; Westwood, 1978). Earlier spring growth may occur and coupled with longer, milder autumns, which promote strong flower bud development, will extend the growing season but spring frosts will damage premature flowering (Farming Futures, 2008). There are some cider varieties that are now flowering one week earlier than they did in the 1970s (Farming Futures, 2008). Fruit growth will be affected more by fluxes in temperature early on in the season than later on in the season (Caldarón-Zavala, 2004). It is predicted that there will be an increase in the number of very hot summer days, also resulting in the growing season lengthening, although the soil moisture levels in the summer and autumn are expected to decrease and there may be heat and/or drought stress reducing yield and fruit or juice quality (Beckenham, 2009; Copas *et al.*, 2008; Farming Futures, 2008). If no drought stress is experienced, then the warmer summer weather with good light levels will increase fruit sugar content and result in a higher quality juice (Farming Futures, 2008). Prolonged hot, dry weather over 20°C, if experienced, could lead to root stress, defoliation, sun scorch, premature fruit drop, poor fruit size or damaged and unmarketable fruit (Farming Futures, 2008). Hot

summer weather is not guaranteed and may be extremely unpredictable as seen in 2007, with high levels of rain and cloud cover, leading to poor photosynthesis, nutrient leaching, small fruit with low juice sugar content, increased levels of scab, increased levels of rotten fruit, making storage a problem and premature fruit drop (Farming Futures, 2008).

Autumn storms may result in branches breaking, loss of immature fruit and disrupt the harvest (Farming Futures, 2008). Heavier winter precipitation is expected to become more frequent and this can lead to waterlogging (depending on soil type) causing root death or crown rot caused by *Phytophthora* spp. (Copas *et al.*, 2008; Farming Futures, 2008). Apple trees require several weeks of relatively cold weather to complete dormancy, their respiration is at its lowest level during the winter, but winter storms and mild, wet and windy weather are likely to become more frequent in incidence (Copas *et al.*, 2008; Farming Futures, 2008; Wibbe *et al.*, 1993). Inadequate dormancy will have different effects on different apple varieties, with some flowering too early risking late frost damage, some flowering at a different time from that required by their pollinator leading to pollination problems, “blind buds” occurring which fail to develop and poor leaf quality leading to a poor fruit set (Farming Futures, 2008). Wind can cause storm damage, stunted growth and reduce the yield (Copas *et al.*, 2008). Among the positive effects, there may be a wider range of crops as climate determines the limits of species distribution and the areas that commercial tree fruit production can occur, but climate change may extend the range (Beckenham, 2009; Pereira, 1975; Westwood, 1978).

Productivity of the orchards may be affected as the levels of CO₂ increase. It is known that as the levels of CO₂ rise, there is more CO₂ available for tree growth, however it is unclear whether this will have an effect on fruit yield (Farming Futures, 2008).

The changing climate will also have an effect on the C balance (C sequestered against that lost in respiration) of orchards as temperature is the predominant factor which influences the carbon balance before and after leaf fall (Wibbe *et al.*, 1993). Although the exact rate and implications of the projected climate change parameters are unknown, the industry is aware of the potential risk and tree growth, thus yield, and

fruit quality will be affected as the climates alter from the ideal of sheltered and sunny (Copas *et al.*, 2008). Warm temperatures and a late leaf fall produce large photosynthetic C assimilation with faster respiration rates during and after leaf fall, resulting in an overall positive C balance (Wibbe *et al.*, 1993).

The World Apple and Pear Association (WAPA) have predicted that the 2012 apple and pear production in the European Union (EU) will be significantly lower than average, with apple production decreasing by 9%, due to the unfavourable climatic conditions in winter and spring across Europe (WAPA, 2012). The climatic conditions are particularly crucial during blossom and the beginning of the growing period as this will have a significant effect on production values. The 2012 European crop forecast for apples is announced at 9.1 million tonnes, which is 7% lower production than the average for the last three years (WAPA, 2012).

1.3 The UK apple industry and market opportunities

As evidence that eating fresh fruit and vegetables may have a positive effect on health increases, it is encouraging that fresh produce, fruit and vegetables, are an important part of the average person's diet accounting for 14.8% of the UK weekly expenditure on food and drink in 2005 and is one of the fastest growing sectors in food and drink retail (Beckenham, 2009). In 2007, according to the Defra Orchard Fruit Survey (Defra, 2007), there were 735 thousand tonnes of apples in the UK annual supply; 33% of this was home production marketed (HPM) and the remainder were imported apples. The UK imported approximately £319,000,000 worth of apples into the country, whilst exporting around £12,000,000 worth (which equates to the equivalent of about 30 thousand tonnes) (Defra, 2007). Approximately £3.6 billion was spent on fruit alone by the UK public in 2007 and the UK market for fresh produce rose by 21.7% from 2001 to 2005 (Beckenham, 2009).

The total area of planted fruit in the UK was 27,580 ha in 2007, with orchard fruit, including apples, pears, plums and cherries, accounting for 59% and the remainder being soft fruit, such as raspberries, strawberries and blackcurrants (Defra, 2007). Apples are

of significance as shown in the 2007 survey, where they made up 48% of the total fruit area planted in the UK with 8,670 ha (Defra, 2007).

The UK grows a variety of dessert, culinary and cider apples (Food Standards Agency, 2006). Total orchard area in the UK is reported to have declined 3% in 2007 from 2004, with the area of dessert apples declining 20% in total and culinary apple area being 5% smaller than the previous survey (Defra, 2007). However, the area of cider apples and perry pear orchards increased by 21% (Defra, 2007). In 2007, there were 4,590 ha of bush cider apples planted, which is a similar level to that recorded in the previous 6 years, and 1,681 ha of traditionally planted cider apples remain (Defra, 2007). The follow up survey published in 2009 reported a slight increase in apple orchard area by 1.7% since 2007 (13,594 ha total planted area), with cider apple varieties continuing to make up the majority with an area covering 6,624 ha. In 2009, a report was published by the Horticultural Development Company (HDC) reporting the results of a survey of core horticultural activities of the sectors paying a statutory levy for near-market research and development (Beckenham, 2009). According to this report, the UK has 152,900 ha of fruit and vegetable production area with commercial orchards accounting for 20,800 ha (13.6%) of the total and producing 284,000 tonnes of orchard fruit (apples, pears, cherries and plums) per annum in 2006 (Beckenham, 2009).

Defra (2007) state that the regions of the West Midlands, Eastern, South East and London and the South West have 98% of the total area of commercial orchards in England and Wales, with the South East region having the largest area of the four. It is thought that the best areas within the UK for growing cider apples are found within the western regions of England, where there is suitable soil and climate for ideal cider apple growth. This is confirmed by the Defra Orchard Fruit Survey (2007), which found that the West Midlands and South West regions have 94% of the total area of cider apples and perry pears, whereas the Eastern and South East regions have 87% of the total area of dessert apples and pears (Beckenham, 2009).

The main focus of this study is on the varieties Cox and Bramley seedling as currently they are the two most widely planted varieties in the UK. Approximately 44% of the

4,810 ha total area of dessert apples planted is the Cox variety (Defra, 2007). According to English Apples and Pears (2012), approximately 60% of the total volume of the commercial production of eating apples grown in the UK is the variety Cox. Bramley, which is only grown in the UK, is the most popular culinary variety, making up 95% of the area planted and representing more than 90% of cooking apples grown in the UK (Defra, 2007; English Apples and Pears, 2012). Although Bramley and Cox are still the most widely planted culinary and dessert apples respectively, the area planted in the UK of these varieties is decreasing due to the increasing popularity of new varieties. In 2007, Bramley had declined 4% and Cox by 32% since 2004 (Defra, 2007).

The decline of dessert apples is largely due to the decrease in area of Cox apples, which has fallen by over 1,000 ha since 2004, and although Braeburn has increased by 40% and Gala has increased by 11%, all the other major dessert varieties have fallen. Other orchard fruits have decreased too, with area of pears down by 8% and the area of plums down by 15%. The orchard area across England has declined by 63% since 1950, from 108,555 ha recorded by the agricultural census to the current Ordnance Survey figure of 39,600 ha (Robertson, 2009). However, the area of cider apples and perry pears has increased since July 2004 by 21% and the area of cherries has increased by 15%; Defra, 2007).

UK self-sufficiency has declined over the past 10 years with a reduction in planting area and a fall from approximately 55% of fruit and vegetables consumed between 1988 and 1993 being domestically sourced, to just 33% in 2006 (Beckenham, 2009). According to Borrie and Potter (2005), the UK is only about 8% self-sufficient in fruit. Planting area for apple production decreased by 33% between 1997 and 2006, while volume of production actually rose by 29%. The number of registered apple growers within the UK has steadily declined from 1,500 in 1987 to just 500 in 2003. Pear and plum production area decreased by 40% and 31% respectively, while pear production volume decreased by 14% but plum production volume increased by 17% (Beckenham, 2009). Although land area of production has decreased, the volume produced has increased as a result of new technology, better yields and cost reductions, the more efficient method of planting commercial orchards. This suggests that production levels of indigenous produce could

be significantly increased within the next 20-30 years (Beckenham, 2009; Common Ground, 1989). This decrease in self-sufficiency has led to an increase in imported fruit and vegetables, which has an impact on the UK economy and environment. Consumers are demanding products that are available year round, despite the constraints of growing season and there is an increased interest in the more variety that non-indigenous produce brings (Beckenham, 2009). The UK apple market is dominated by varieties grown in all the major production areas of the world, which enables a constant fruit source for customers over most of the year (Food Standards Agency, 2006). The climate and growing season imposes constraints on UK production of crops, however 20 years ago there was a 22% higher domestic production, providing evidence that there is a change in consumer demand towards imported goods. Early UK varieties may be marketed immediately after picking but most go into cold storage for up to 12 months prior to marketing (Food Standards Agency, 2006). Due to advances in storage technology since the 1950s, fruit can be held in good condition in refrigerated cold stores with atmospheric control of the oxygen (O₂) and CO₂ levels (Food Standards Agency, 2006).

Government statistics show that currently an average of 3.9 portions per day are purchased by each household (3.5 by those with a low income). However, this is based on purchasing figures and actual consumption may be lower as it is estimated that more than 10% of purchased household food is lost as waste (Beckenham, 2009). Thus, there is a poor level of fruit and vegetable consumption in the UK population with approximately 250 g per day instead of the recommended 400 g per day (Beckenham, 2009). There are schemes set up by local councils to try to encourage more children in the UK to eat fresh fruit and vegetables regularly. One such scheme is the UK School Fruit and Vegetable Scheme, where children aged 4 to 6 in Local Education Authority (LEA) maintained schools are entitled to a free piece of fruit every day (Borrie and Potter, 2005). More recently, the European Parliament adopted a similar scheme in a bid to combat the rising epidemic of poor health and obesity within children throughout the UK and Europe (nearly one in three 10-11 year olds in the UK is overweight). A report in 2007 found that the UK School Fruit and Vegetable Scheme successfully increased five-a-day consumption in school children from 32% in 2004 to 44% in 2006 and it is hoped

that an EU-wide scheme will help to improve existing schemes and stimulate new initiatives by providing a formal network (Fresh Produce Consortium, 2008).

There are organisations in place, both in the UK and world-wide, concerned with the marketing of fresh fruit. In the UK, the “umbrella” organisation for UK marketing organisations is English Apples and Pears Ltd (Food Standards Agency, 2006). The organisation formed in 1990 performs various industry wide functions and aims to organise and develop the promotion of the English Industry (English Apples and Pears, 2012; Food Standards Agency, 2006). They represent England in the WAPA, act as a trade association, develop public relations activities, such as the Bramley campaign and safeguard the interests of its members whilst promoting English grown fruit (English Apples and Pears, 2012; Food Standards Agency, 2006). WAPA (2012) was established in 2001 as an industry body to represent major apple and pear producing countries globally. Some of the work carried out by the body includes the improvement of producing countries’ business activities, increasing the demand for apples and pears and matching supply and demand, whilst ensuring a fair return for producers. Market information is analysed and disseminated, which enlarges the producing countries’ knowledge base and allow the members to react to changing market trends. In the UK, for growers who prefer to market their own fruit directly, the British Independent Fruit Growers Association (BIFGA) is a trade association to protect their interests (Food Standards Agency, 2006).

1.4 UK orchards

An orchard can be defined as a piece of land with two or more fruit trees growing on it that are producing fruit and have a variety of purposes, values and benefits to the community and its wildlife, flora and fauna (Common Ground, 1989). They are found on a range of sites from level to steep slopes, with any aspect and generally good drainage, although this varies (Robertson, 2009). Orchards are found throughout the lowlands of the UK (Figure 1.2) and can be categorised as either traditional orchards or commercial orchards.

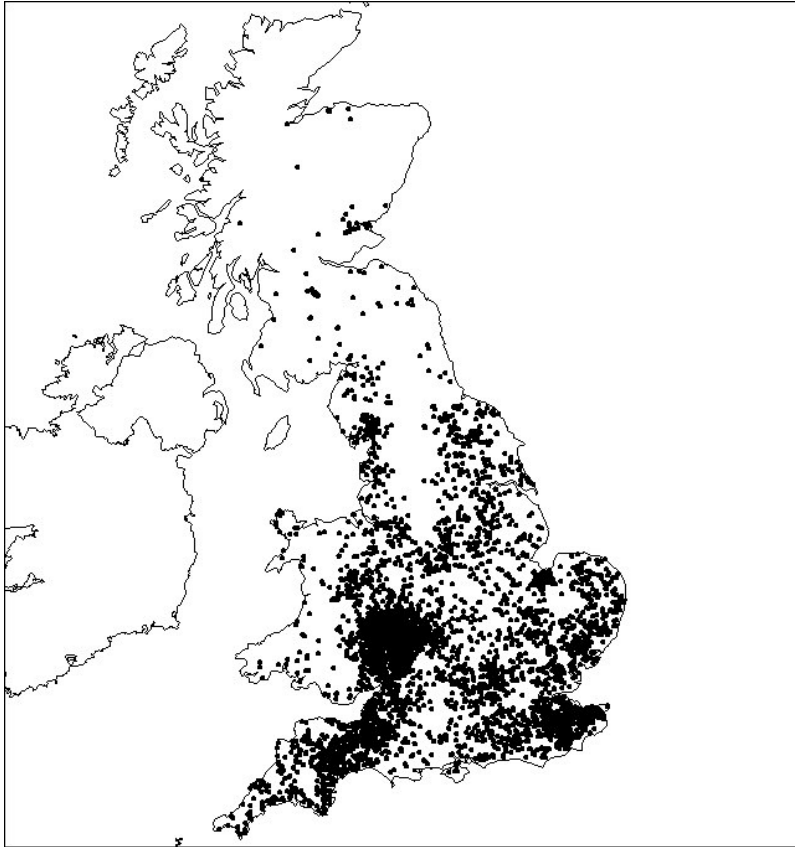


Figure 1.2. Orchard distribution, both traditional and commercial, throughout the UK in 2007 (where each dot represents one orchard). This map is reproduced from Ordnance Survey material with the permission of Ordnance Survey on behalf of the Controller of Her Majesty's Stationery Office © Crown copyright.

1.4.1 Traditional orchards

Traditional orchards consist of fruit and nut trees planted at low densities in permanent grassland with low intensity management, where chemical pesticides and fertilisers are rarely used (Natural England, 2010a). Widely distributed across the UK, traditional fruit orchards include apple, pear, cherry, plum and damson trees. Sheep grazing can occur in these types of orchards, which are usually found on a wide range of soil types from slightly acid, infertile soils to fertile river floodplain soils and lime-rich soils (Robertson, 2009). They have positive social, environmental and economic impacts by providing a highly appreciated landscape, important for local community use, preserving living collections of historical fruit varieties, producing cider, perry and fruit juice to sell whilst providing a range of ecosystem services (Common Ground 1989; Robertson, 2009). These ecosystem services include an amelioration effect of climate change via being a C sink and providing important habitats for wildlife. Traditional orchards provide a mosaic of different habitats, with the most important being the individual trees and their

associated habitats, including scrub, hedgerows providing berries as food, unimproved grassland rich in wildflower species, fallen dead wood, which is habitat for insects and invertebrates such as the stag beetle, *Lucanus cervus*, and ponds (Natural England, 2010a; Robertson, 2009). Due to the conservation importance of traditional orchards and their total area declining in recent years, they have been classed as priority habitat in Wales under section 42 of the Natural Environment and Rural Communities Act (2006) and area a national priority in the UK Biodiversity Action Plan (BAP) (Joint Nature Conservation Committee, 2008; Natural England, 2010b). Traditional orchards support a wide range of wildlife and provide BAP priority habitats and species, whilst offering Nationally Rare and Nationally Scarce species, (including four priority BAP beetles such as the scarce orchard noble chafer beetle, *Gnorimus nobilis*, which is a priority BAP species almost entirely restricted to traditional orchards, Natural England, 2010a; 2010b). There are over 400 specialist wood-decay species that have been recorded in traditional orchards, including 102 Red Data Book or Nationally Scarce species (Natural England, 2010b). Traditional orchards have high biodiversity value and offer valuable habitats to local wildlife; animals, birds, insects and plants, whilst also producing increasingly rare and invaluable varieties of fruit (Common Ground, 1989; CFWI, 1995; Robertson, 2009). In summer, the leaf canopy provides nesting sites for birds (such as resident blackbirds (*Turdus merula*), titmice (*Baeolophus bicolor*), mistle thrush (*Turdus viscivorus*), chaffinches (*Fringilla coelebs*) and insect-eating summer migrants such as the garden warbler (*Sylvia borin*), whitethroat (*Sylvia communis*), willow warbler (*Phylloscopus trochilus*) and blackcap (*Sylvia atricapilla*)). The fruit itself is an important food source for birds (e.g. thrush, *Turdus* sp.) and butterflies (e.g. the Red Admiral, *Vanessa atalanta*, like the decaying fruit) and the crab apple, *Malus sylvestris*, is associated with 93 different species of insect (CFWI, 1995). Not only do birds and insects take advantage of fallen fruit in the winter, but it also provides forage for mammals such as the badger (*Meles meles*), fox (*Vulpes vulpes*), mice (*Mus* spp.), voles (*Microtus agrestis*) and hedgehogs (*Erinaceus europaeus*) (CFWI, 1995).

Traditional orchards provide habitat for epiphytic flora and fauna, with 16 Nationally Rare or Nationally Scarce species of lichens being recorded, including a very rare and protected species *Parmelinopsis minarum* (Natural England, 2010b). The rare BAP

species, the orchard tooth fungus *Sarcodontia crocea*, has been found at only 15 sites in the UK and was found only on apple trees (Natural England, 2010b). A good variety of barkflies (*Psocoptera spp.*) and the Nationally Scarce apple-tree lace bug, *Physatocheila smreczynskii*, are found within UK traditional orchards (Natural England, 2010b).

The key to maximising the wildlife value of an orchard is low intensity management without the use of chemicals (Natural England, 2010a). If an orchard is managed or restored in this way, it is possible to achieve a reasonable yield of fruit, preserve habitats and reinforce the landscape (Natural England, 2010a).

1.4.2 Commercial orchards

Commercial orchards aim to produce maximum fruit yield possible within the annual production year; they comprise cider and culinary types and consist of smaller trees (approximately 1.8 to 2.4 m tall) planted densely as bush or cordon varieties in rows for ease of mechanical operations, such as spreading fertilisers and pesticides (Common Ground, 1989; CFWI, 1995). This planting density is more economically viable. Tree varieties that are chosen for commercial orchards are those with sales appeal and which produce uniform fruits, which develop at the same rate and can survive mechanical harvesting without too much damage or bruising (CFWI, 1995). Cider apple cultivars grow too vigorously on their own roots, which delays cropping. Intensive bush orchards bud the scion cultivar onto a semi-dwarfing rootstock which ensures a greater uniformity between trees in term of growth and habit and improves the cropping (Umpelby and Copas, 2002). Commercial apple trees are usually replaced every 12 to 15 years due to loss of economic value and increased disease pressure (Food Standards Agency, 2006). Commercial orchards still offer a habitat to wildlife and many of the insects, birds and animals associated with traditional orchards can be found within the more intensely harvested orchards (Common Ground, 1989).

Traditional orchards are planted at approximately 150 trees ha⁻¹, while commercial orchards are much more densely planted with approximately 2,200 trees ha⁻¹ and are usually planted on dwarf rootstock (Local Habitat Action Plan for Cambridgeshire, 2003). Within both traditional and commercial orchards, the role of bees (*Apis mellifera*) is vital

as they pollinate the flowers of the trees in spring and ensure a good crop later in the year (Common Ground, 1989; MAFF, 1958). Orchards and beekeeping often go hand in hand as the blossom of the fruit trees provide an important source of pollen for bee colonies, which enables them to build up large populations and produce surplus honey in time (Common Ground, 1989). Bees do not fly well in rain, strong wind, or at temperatures below 10°C (Westwood, 1978). It has been observed that the changing climate has already had an effect on the number of bees in the UK, with nearly one in three of Britain's honeybee hives failing to survive during the winter and spring of 2007 due to bad weather, disease (such as the varroa mite originating from southeast Asia) and changes in farming methods (BBKA, 2009;). A decline in UK beehives will have an effect on UK orchards in terms of pollination, potentially leading to severe consequences in terms of fruit yield for apple growers due to the fact that 90% of apple trees are pollinated by bees in the spring. Other fresh produce from the UK will be affected, such as pears, raspberries, cucumbers and cherries, as bees pollinate a third of the total produced (Defra, 2009). Defra and the Welsh Government published "Healthy Bees" in March 2009, which is a ten year plan to protect and improve the health of honey bees in England and Wales (Defra, 2009; The Fruit Grower, 2009). The investment of an extra £4.3 million to gather information and research bee health aims to sustain honey bee populations by supporting beekeepers to minimise the risk from pests and diseases, whilst initiating a research programme on pollinators (Defra, 2009; The Fruit Grower, 2009).

1.5 Crop description

The domesticated apple fruit, (*Malus x domestica* Borkh.), is the main fruit crop of temperate regions of the world and the most important world fruit crop after oranges, bananas and grapes, with world production being estimated at 56 Mt in 1998 (Jackson and Looney, 1999; Velasco *et al.*, 2010). Apples form a significant part of the UK diet and are the most important tree fruit grown commercially in the UK at this present time in terms of area cropped, volume and value (Food Standards Agency, 2006). The cultivated apple belongs to the genus *Malus*, which comprises some twenty five to thirty five species along with several subspecies distributed almost continuously throughout temperate Eurasia and North America (Food Standards Agency, 2006). Although still of

importance today, they have been grown for thousands of years (Food Standards Agency, 2006). They are thought to have originated in Western Asia, Eastern Europe, and southwestern Siberia, with several crab apples and other *Malus* species being native to China, India and North America (Westwood, 1978). The primary centre of diversity for the *Malus* is in the region of Asia Minor and Western China (Food Standards Agency, 2006). China now produce approximately one-third of the world's apples due to large-scale planting during the 1980s and the 1990s (Jackson and Looney, 1999).

Today, wild apples are found throughout Europe, the Caucasus region, middle Asia and China. The common apple of Europe is the domesticated apple *Malus pumila*, one of the hardiest of temperate zone fruits, and is thought to have derived from complex hybrids of several wild species (Jackson and Looney, 1999; Westwood, 1978). Several thousand cultivars of apple exist, but there is a tendency towards concentrating commercial operations on a reduced range and a smaller number of marketable apples with widely accepted taste and texture. Consequently, a relatively small number of cultivars account for most of the world's apple production (Jackson and Looney, 1999; Westwood, 1978). One such example is the English Cox, nearly 82 million kg of which are harvested annually in the UK (Copella Fruit Juices, 2008). The apple has a wide diversity in climatic adaptation, hardiness, maturation length and this wide genetic diversity has enabled it to become the most widely planted fruit in the temperate zone (Westwood, 1978). Apples are now grown in orchards with known varieties and are grown on specifically selected rootstocks for agronomic qualities such as fruit size control, tolerance of certain soil conditions and pest and disease tolerance (Food Standards Agency, 2006). Varieties are selected for their marketable traits, including fruit quality, eating attributes, storage potential alongside valuable agronomic traits, such as yield, fruit size distribution and the tolerance of the variety to pests and diseases (Food Standards Agency, 2006).

Apples (*Malus* spp.) are termed pome fruits, along with pears (*Pyrus* spp.), quince (*Cydonia* spp.), service berry (*Amelanchier* spp.), hawthorn (*Crataegus* spp.) and rowan (*Sorbus* spp.) and are from the family *Rosaceae*; subfamily *Pomoideae* (Jackson and Looney, 1999). The genus consists of about fifteen primary species, two from Europe, four from North America and the rest from Asia (Westwood, 1978). Pome fruits are

commercial fruit species and formed by a fusion of the ovary and receptacle, while multiple seeds are borne in five carpels consisting of mesocarp and exocarp tissues (Jackson and Looney, 1999). *Malus* spp. are deciduous, rarely evergreen trees or shrubs with rarely spiny branches, ovoid buds, serrate or lobed leaves, white to pink flowers that have usually suborbicular or obovate petals, containing 15-50 stamens usually with yellow anthers. The ovary is inferior with 3-5 cells and there are 2-5 styles (Westwood, 1978). Mostly apples are consumed as a fresh product although there are other forms of consumption, thus there are four basic types of apples: cider (for making cider), crab (ideal for making jam or jelly), dessert (eating apples) and culinary (which are apples for cooking) (Hessayon, 1990; Jackson and Looney, 1999).

Malus spp. like well-drained soil and are grown under cool or warm temperate conditions at latitudes of 35-55°, where they thrive in full sun and can receive sufficient winter chilling, but not freezing as damage to the tree can occur at temperatures below -20°C (Hessayon, 2003; Jackson and Looney, 1999; Westwood, 1978). When air temperatures reach 35-40°C, apple fruits become susceptible to sunburn and can suffer heat damage. The surface that is exposed first turns pale, then brown, followed by a black colour with fungal infection occurring in serious cases. Low volume misting sprinkling can prevent this from occurring (Jackson and Looney, 1999). They do grow well in areas that have salt-laden sea air or shallow alkali soil and benefit from shielding in windy areas (Hessayon, 1990). Optimal growing conditions include between approximately 500 - 1000 mm rainfall per annum, altitude less than approximately 150 m and soils that are not too nitrogen rich ideally with pH range 5.5-6.5, although culinary varieties grow more successfully than dessert trees in less than optimal conditions (Hessayon, 1990; Jackson and Looney, 1999; Westwood, 1978). Frosts and temperatures below -2°C can damage or kill flowers and young fruit after open cluster, if the frost is only light, it still may cause damage to the fruit tissue and result in marking or deformities that persist until maturity (Jackson and Looney, 1999). It can be very damaging to apple trees if there is a period of mild weather in the winter followed by a period of very low temperatures. Very low night temperatures, followed by exposure to a hot sun, may cause the bark to split and sun scald wounds on the southerly side of the trunks and branches, Cox's Orange Pippin is particularly susceptible to this type of

damage (MAFF, 1958). *Malus* spp. are reasonably tolerant of drought or wet soils, dependent upon which rootstock it is grafted onto, and need to have water supply maintained to the tree roots as a lack of water results in a reduced yield (Jackson and Looney, 1999; MAFF, 1958). Excess water may produce fruit with a poor lasting ability. If the leaves of the trees remain wet for long periods due to prolonged, frequent rain, it is difficult to prevent the tree from becoming disease ridden (MAFF, 1958). High humidity increases the likelihood of diseases such as scab and Nectria canker (Hogg, 1975; Jackson and Looney, 1999). The main pests and pathogens of *Malus* spp. are apple scab, powdery mildew, codling moth and red spider mite (Jackson and Looney, 1999). Average annual rainfall of between 500 and 1000 mm is required for the production of good quality apples (ideally for cider apples, this should be around 400 mm during the summer months, from April to October) and any amounts over that, severely limits the chance of success (MAFF, 1958; Umpelby and Copas, 2002). Excessive nitrogen fertilisation should be avoided as it encourages vegetative growth, whilst reducing storage quality and colour in red varieties (Jackson and Looney, 1999).

Apples do not grow well on their own roots and they are often grafted on to selected rootstocks, which provide a root system, and which influences the vigour of the apple tree and will determine the eventual height of the tree (CFWI, 1995; Hessayon, 1990; Hessayon, 2003; Westwood, 1978). Grafting has been used for fruit tree propagation for over 2,000 years and is the exploitation of cambial cells, which are placed in close contact and firmly held together until a callus is formed, which unites the graft and new vascular tissues differentiate (Cannell and Jackson, 1985; CFWI, 1995). Rootstocks that are used for grafting onto apple trees, for optimal growth, were developed at East Malling Research Station (the M series) and Merton (the MM series) (CFWI, 1995; Hessayon, 1990; Jackson and Looney, 1999; Westwood, 1978). A wide range is used including: very dwarf (M.27), dwarf (M.9), semi-dwarf (MM.106, M.7), standard (M.793) and large (MM.115, M.25) (CFWI, 1995; Jackson and Looney, 1999). A mature dwarf bush will grow to approximately 1.8 m high and yield about 0.02 tonnes of fruit annually, while a mature standard tree will grow up to 7.6 m and yield between 0.1 and 0.2 tonnes of fruit per year (Hessayon, 1990). The expected yield varies depending on the site and the cultivar and rootstock combination, but yield will also vary between

intensive, more densely planted, orchards, achieving 20 t ha⁻¹ after just 3 years and up to 70 t ha⁻¹ after 8 years, and semi-intensive orchards, with lower planting densities per hectare, achieving 10 t ha⁻¹ after 4 years and taking 10 years to yield up to 70 t ha⁻¹, thus an intensive orchard tree will reach full production in 5-8 years, with a semi-intensive orchard tree reaching full production in 8-10 years (Jackson and Looney, 1999). Cider apple trees can yield up to 0.5 tonnes of apples per tree annually (Hessayon, 1990).

There are a number of advantages to grafting onto dwarfing stock; more flower buds form on young trees and therefore fruit more quickly, the fruit is more brightly coloured, picking is easier and pruning becomes much less complex (Hessayon, 1990; Jackson and Looney, 1999). However, dwarf stock requires a support system (e.g. trellis), has a more confined root system, whilst requiring fertile soil, regular watering and producing a lower yield, therefore a compromise must be reached and semi-dwarfing stock or even vigorous stock should be used for planting in poor soil types (Hessayon, 1990; Jackson and Looney, 1999).

Apple trees are insect pollinated, mainly by honey bees, but there are a few apple varieties that are self-fertile, such as *James Grieve* and *Arthur Turner*, which means they have the capability to set some fruit with their own pollen. However, it is necessary to have a pollination partner nearby in most cases (Hessayon, 1990; Jackson and Looney, 1999; MAFF, 1958; Westwood, 1978). Pollination is the transfer of pollen from the anther to the stigma and after pollination, the pollen grain germinates, grows a tube that extends down the style and fertilisation occurs once the male nucleus from the pollen tube unites with the egg cell in the embryo sac to form a seed (MAFF, 1958; Westwood, 1978). A pollination partner is a tree of another variety which flowers at approximately the same time (and is in the same flowering group or the next immediate group), thus enabling cross-pollination to take place. There are four flowering season groups for apple classification: early flowering, mid-season flowering, mid-season/late flowering and late flowering (Hessayon, 1990). There are varieties which are very poor pollinators, called the triploid variety, and it requires two non-triploid varieties to be grown in the vicinity of the triploid tree to act as pollination partners (for both the triploid variety and each other) (Hessayon, 1990; MAFF, 1958). Flowering occurs in

spring and flower buds are produced on tips of shoots or on spurs of wood that is 2 years old or older (Jackson and Looney, 1999; Westwood, 1978). The buds are vegetative and mixed, with 5 to 8 flowers, which each contain 5 sepals, 5 petals, 5 pistils and approximately 20 stamens (Jackson and Looney, 1999; Westwood, 1978). Flower buds are usually borne terminally on shoots or short spurs, but can be borne on lateral buds of one-year shoots (Westwood, 1978). Fruit growth follows a sigmoidal growth curve (Jackson and Looney, 1999).

Fruit size is of particular importance commercially and is controlled by providing adequate nutrients and water, removal of weeds to avoid competition, pruning to ensure renewal of fruiting wood and regulating the crop size by thinning (Jackson and Looney, 1999). Thinning levels will depend on the cultivar but it is an important tool to prevent biennial bearing, which is where a large crop one year is followed by a poor crop the following year. Cox's Orange Pippin is an example of an apple variety which is susceptible and displays biennial bearing of fruit. Biennial bearing affects whole trees, whole orchards or even whole districts and involves inadequate flowering and is triggered by an event, such as frost or disease, and is not caused by inadequate pollination (Jackson and Looney, 1999). During the year of the event, flower production is excessive and the following year's crop is so large that it limits flower initiation in the subsequent year after that, as an apple spur that has flowered one year is unlikely to flower the next (Jackson and Looney, 1999). If the apple tree is thinned, where a proportion of the fruit is removed, within 3-4 weeks of full bloom, the biennial bearing will be affected significantly and the most economic method of doing this is to use a chemical thinner. As individual spurs age, they become increasingly more biennial, therefore pruning is essential to maintain relatively young fruiting wood and to reduce the susceptibility of the fruit trees to biennial bearing (Jackson and Looney, 1999). Pome fruits should be pruned lightly to develop a spur system, which bear the fruit crops, heavy pruning results in the growth of many long, unfruitful shoots, which would have remained short and fruitful (Westwood, 1978). Pruning directs the growth of the tree, regulates the number and position of fruits buds that develop and aims to remove weak, thin and unproductive shoots, whilst thinning out areas on the tree that are too dense and inhibiting light distribution (MAFF, 1958; Westwood, 1978).

Apple trees can live for over 100 years, examples of these can be found in traditional orchards, with normal fruit production life being up to 40 years (Food Standards Agency, 2006; Jackson and Looney, 1999). However, as fruit quality and yield decline as trees age, commercial orchards are often replaced much sooner with more profitable cultivars as the orchard has an economic life of 12 to 15 years (Food Standards Agency, 2006). Most commercial orchards are harvested by hand and apples are then stored in the cool and under controlled atmospheric conditions. These controlled conditions allow the apples to be stored for up to a year so that they are in perfect condition when the market demands them. Transfer from orchard to storage must happen quickly, as any delay may cause several days of reduction in effective storage life (Jackson and Looney, 1999).

1.6 Background of research

The management of C stored in land-use systems is a major objective with the Kyoto Protocol (Article 3.4) as a method of mitigation to meet C emission targets (Smith *et al.*, 2000; Swingland, 2003). Temperate woody ecosystems, such as traditional orchards, store large pools of C in the soil, which exceed the C store in above ground biomass. A preliminary analysis of Herefordshire orchards (Khan, 2008) showed that extensive orchards have higher soil C stocks several fold higher than intensively managed orchards, with approximately 30 t C ha⁻¹ compared to 10 t C ha⁻¹ respectively in the top 30 cm depth of the soil profile. It was found that the C stored in the above ground biomass of the large tree orchards also exceeded that of the densely stocked bush orchards (Khan, 2008). These data suggest that the amount of C stored in the system is strongly affected by the orchard management.

1.7 Research objectives

This study aims to determine the C storage potential of orchards in relation to the management system, and to estimate the environmental value of C stored. This will be achieved by focussing on several specific objectives:

- Determine the C stored in above and below-ground biomass and soils in different orchard management types
- Determine the C sequestration of different orchard management types

- Calculate the C footprint of different orchards, with standard management data on inputs and outputs for real orchard ecosystems

Project overall hypothesis:

There will be more C stored in the above- and below-ground biomass and the soil of older orchards planted under the traditional planted method.

1.8 Thesis structure

This thesis is divided into 7 chapters:

Chapter 2: Quantifying carbon (C) in UK apple orchards

This chapter quantifies the C found within the above- and below-ground biomass of *Malus* trees and that found within the soil in comparison to that of other land uses. Height and diameter at breast height (dbh) measurements are recorded and modelled according to two allometric equations to determine C storage in the biomass and roots, while the analysis of soil samples quantifies soil C content. Apple variety and orchard age are explored as factors of C content.

Chapter 3: Soil carbon (C) fluxes in response to the grubbing-out management practices in UK apple orchards

In response to the management practice of removing whole trees including roots and stumps (grubbing-out) when the *Malus* trees, the responses of the orchard in terms of C lost are investigated. Measurements of the CO₂ released from the system and their analysis are presented to determine the amount of soil C stocks being depleted by this action and what proportion is trapped CO₂ being released.

Chapter 4: Biochar-mediated changes in soil quality in a Bramley apple orchard pot trial

The potential of using *Malus*-derived biochar as a soil amendment in orchard systems is determined by quantifying soil quality changes in order to improve knowledge on the effects of using charcoal in this manner over a long-term study. The results of a pot trial carried out with Bramley (n = 17) apple trees planted in four treatment types (0, 5, 25

and 50 t char ha⁻¹) are presented. Throughout the experiment soil respiration, tree height and dbh are recorded and soil properties are quantified to determine any changes in soil quality.

Chapter 5: Evaluation of the cell wall properties and microstructure of *Malus*-derived biochar

The properties of *Malus*-derived biochar, such as surface area, density, pore size distribution and volume are characterised in this chapter. This information will improve understanding of the absorptive properties of biochar and the implications of its burial in the ground.

Chapter 6: Evaluation of environmental impacts of greenhouse gas emissions of apple production from UK apple orchard systems

With increasing emphasis on reducing GHG emissions, it is important to determine the impact of emissions from the food industry as little is yet known about these systems. This chapter quantifies GHG emissions from UK apple orchards, with the focus on Bramley and Cox varieties, and one example of a Gala and a Rubens orchard. Potential areas in which GHG emissions could be reduced are identified, which will inform both on-farm management choices and aid policy makers.

Chapter 7: General discussion and conclusion

The main findings of the results chapters are drawn together and the overall thesis aims are addressed. Conclusions are drawn from the study on the global implications of UK apple orchards and the environment, while further work is identified.

CHAPTER TWO

QUANTIFYING CARBON (C) IN UK APPLE ORCHARDS

Abstract

Due to increasing concern regarding the changing climate, there is an interest into the understanding of land use effects and management of ecosystems in terms of carbon (C) storage. Temperate woody ecosystems are potential C sinks, storing C in the soil, below- and above-ground biomass. The aim of the study was to determine the amount of C stored within UK Bramley and Cox apple orchards of differing management types and that of other habitats for comparison. Soil samples ($n = 10 \text{ ha}^{-1}$) were collected to a 5 cm depth from both Cox and Bramley orchards of two age categories (< 15 and > 15 years old), traditional cider orchards ($n = 3$) and from various surrounding land uses for comparison. Soil was collected from one site to a 100 cm depth in order to determine C content over the soil profile. Tree height and dbh were recorded ($n = 40$ per orchard) and analysed by allometric equations to determine C in the above- and below-ground biomass. Significantly ($p < 0.001$) higher soil C was found between age categories, with means of 7.02, 12.63 and 16.21 t C ha^{-1} (< 15 , > 15 and 35+ age groups respectively). The apple variety also influenced the amount of C stored, with Bramley soil having significantly higher ($p = 0.005$) C (mean $< 15 = 8.86 \text{ t C ha}^{-1}$, mean $> 15 = 14.33 \text{ t C ha}^{-1}$) than that found under Cox orchards (mean $< 15 = 7.60 \text{ t C ha}^{-1}$, mean $> 15 = 10.34 \text{ t C ha}^{-1}$). The tree and root C followed a similar pattern with significantly ($p < 0.001$) higher C content with increasing age and in that of Bramley orchards than that of Cox orchards (above-ground; $p < 0.001$, Bramley mean = 10.57 t C ha^{-1} , Cox mean = 4.99 t C ha^{-1}). Comparisons between orchards and surrounding habitats showed that orchards store significantly higher C than arable land but they store significantly less C than woodlands. This could be due to the smaller biomass of commercial orchard trees than those found in woodlands and that some of the C in orchard trees is attributed to fruit production. It is concluded that orchards have the potential to store large amounts of C depending on management type in comparison to other land uses, although this C store may be lost when the tree productivity declines and orchards are grubbed out.

Key words: *above-ground biomass, below-ground biomass, climate change, carbon accounting, carbon sequestration, fruit, fruit production.*

2.1 Introduction

2.1.1 Global climate changes

Rising concerns about climate change, along with an increase in environmental and political awareness about rising concentrations of greenhouse gas (GHG) emissions are stimulating research into potential mitigation options. Mean global atmospheric carbon dioxide (CO₂) levels have increased at a progressively faster rate each year from 280 ppm in the 1700s to 380 ppm in 2006, with a recent rapid growth in atmospheric CO₂, up to 2 ppm yr⁻¹ (Canadell *et al.*, 2007a; Raupach *et al.*, 2007). The present concentration is the highest it has been during the last 650,000 years (Petit *et al.*, 1999; Siegenthaler *et al.*, 2005). Under the Kyoto Protocol, the European Union is committed to reduce its carbon (C) emissions by 8% by 2012 compared to baseline levels from 1990 and the UK is committed to a 12.5% reduction (IPCC, 2000; Smith *et al.*, 2000). According to the European Commission (2012), total EU GHG emissions were reduced by 15%, with the largest emitters, Germany and the United Kingdom, achieving a total GHG emissions reduction of 483 million tonnes CO₂e. This reduction within the UK is attributed to the liberalisation of the energy market and the subsequent switch from using oil and coal as fuel to natural gas in electricity production (European Commission, 2012). The management of C sequestration in land-use systems is a major objective with the Kyoto Protocol (Article 3.4) as a method of mitigation to meet C emission targets (Dawson and Smith, 2007; Lal, 2004; Smith *et al.*, 2000; Swingland 2003).

There are three main C reservoirs involved in the global C cycle; the atmosphere, terrestrial biosphere and oceanic biosphere (Schimel, 1995). Studying the C in the terrestrial biosphere is increasingly important in terms of climate change and understanding how the C cycle will be affected as GHG levels rise. It is expected that there will be an increase in the decomposition of organic matter, which will result in an increased release of CO₂ from the soil thus instigating a CO₂ fertilisation effect leading to

an increase in primary production. This in turn will lead to more C returning to the soil (Batjes, 1996).

2.1.2 Determination of C sink or source

C budgeting identifies sources and sinks of C and this identification of C pools and fluxes is important in developing strategies for climate change mitigation (Lal, 2004). The understanding of land use effects on soil C within agroecosystems is required to determine potential options to sequester C and offset GHG emissions (Lal, 2011; Ostle *et al.*, 2009). Between 1978 and 2003, it was calculated that C was lost from soils in England and Wales at a mean rate of $0.6\% \text{ yr}^{-1}$ and the loss was attributed to climate change and substantial changes in land-use and management within the same time period (Bellamy *et al.*, 2005; Dawson and Smith, 2007). Management of ecosystems is the key to whether the system remains a sink of C or becomes a source of CO_2 and methane (CH_4) as changes in the soil organic C content can have an effect on the global C budget (Bellamy *et al.*, 2005; Ostle *et al.*, 2009). The amount of C found in soils under different land use is important when considering managed C sinks as a basis for policy decisions (Howard *et al.*, 1995).

2.1.3 C pools

There are three main C pools which regulate the C cycle worldwide (Table 2.1). The terrestrial C pool, consisting of C stored in vegetation and the soil below, is much more labile than the larger oceanic store of C (Batjes, 1996; Dawson and Smith, 2007). However, this terrestrial store of C is important in the global C cycle as C is retained in the soil, below- and above-ground biomass and decomposing organic matter (IPCC, 2000). It is important to manage the land, such as forests and agricultural areas, in a manner by which increased GHG sequestration can occur and retain the C stored within the biosphere. Human activity, such as land use change, the burning of biomass and environmental pollution, can have a significant impact on the C balance of the ecosystems within the terrestrial biosphere by releasing gases that contribute to the rising global GHG concentration (Batjes, 1996).

Table 2.1. The amount of C stored in the three main C pools which regulate worldwide C cycling according to Batjes (1996).

C pool	C stored (Pg C)
Terrestrial biosphere	Approx. 2200
Atmospheric biosphere	Approx. 750
Oceanic biosphere	Approx. 39 000 x 10 ¹⁵

In the UK, vegetation accounts for a relatively small amount of C stored, approximately 113.8 ± 25.6 Tg (Ostle *et al.*, 2009). Forested land and grassland sequester the most at 110 ± 4 kg and 240 ± 200 kg respectively (Dawson and Smith, 2007; Ostle *et al.*, 2009). Arable and horticultural crops contain approximately a tonne of C per hectare, whereas heath and bog vegetation containing approximately 2 tonnes per hectare (Dawson and Smith, 2007; Ostle *et al.*, 2009). The largest and most key component of the terrestrial C store is the soil, with the total amount of C stored in UK soils being more than twice that held in vegetation at 9.8 ± 2.4 billion tonnes (Batjes, 1996; Bellamy *et al.*, 2005; Dawson and Smith, 2007; Ostle *et al.*, 2009). There are a range of studies, which have calculated various global C estimates within the soil from 1115 to 2200 Pg of C (Batjes, 1992), 1576 Pg C (Eswaran *et al.*, 1993), 1220 Pg C in tropical soils (Sombroek *et al.*, 1993), and between 2157 to 2293 Pg in the upper 100cm (Batjes, 1996). This C store is an important factor to consider under current climatic conditions, as any changes to its abundance and composition will have an effect on the processes taking place within the system (Batjes, 1996). In a UK wide study by Bradley *et al.*, (2005), it was found that the average soil C density found in woodlands was less than that found in seminatural land, with 13 and 15 kg C m⁻² in the top 30 cm of the soil profile respectively, but was higher than the soil C density in arable land which contained 8 kg m⁻². Arable soils usually contain depleted levels of soil organic carbon (SOC), while grassland soil contains 10 – 30% of world soil C (Dawson and Smith, 2007; Scurlock and Hall, 1998). Vanguelova *et al.*, (2013) carried out a survey of the C stocks within UK forest soils and found that there was an average between 108 and 448 t C ha⁻¹ down to 80 cm depth.

2.1.4 Potential of orchards as a C sink

Within plant systems, a C source is usually associated with a respiring tissue, where C leaves the system; while a C sink is when the C import is higher than the export (Blanke,

2009). Temperate woody ecosystems store large pools of C in the soil, which exceed the C store in above ground biomass (IPCC, 2000). Currently, very little is known about the C balance and potential of apple orchards as C sinks (Panzacchi *et al.*, 2012). A preliminary analysis of Herefordshire orchards (Khan, 2008) showed that extensive orchards have higher soil C stocks, in the top 15 cm of the soil profile, several fold higher than intensively managed orchards, containing approximately 30 t C ha⁻¹ compared to 10 t C ha⁻¹ respectively. It was found that the C stored in the above ground biomass of the old tree orchards also exceeded that of the densely stocked bush orchards (Khan, 2008). These data suggest that the amount of C stored in the system is strongly affected by orchard management. Little work has been carried out on the C storage potential of commercial, intensely managed orchards and their impact in comparison to other land-use systems. Although it is known that orchards sequester C, there is limited knowledge on the actual values of the C pools within the system (Warner, 2009). Should orchards be found to have C storage potential, this could be an important driver in determining land-use and land-use change policies. This would be positive for the orchard industry and in terms of climate change, as land use change can result in the rapid loss of soil C (Ostle *et al.*, 2009). With the increasing requirements of C retention in terrestrial systems as a method of mitigation to meet C emission targets, should a C retention value within orchards be found, this would provide strong arguments for land-use subsidy. The draft results from the Herefordshire Orchards Community Evaluation indicate that the value of orchards is much more extensive than currently accredited for, including factors such as C sequestration, flooding alleviation, biodiversity, social cohesion and the local economy, thereby indicating that the potential value of orchards far outweighs the current profit to farmers. A study of Chinese apple orchards showed that between 1990 and 2010, the net C sequestration of the orchards contributed 4.5% of the total net C sink for the terrestrial ecosystems in China (Wu *et al.*, 2012). This study and others suggest that orchard systems have the potential to be net C sinks alongside producing fruit for the food market.

There are a number of potential C stores within an orchard ecosystem; the soil, root biomass and above-ground biomass; including the woody biomass, the leaves and the fruit themselves. The amount of C stored in the soil of a system depends on the amount

of inputs of organic materials and the rate of decomposition (Wu *et al.*, 2012). All of the C pools within a system are interlinked and the soil C will be dependent on the other pools of C within the apple trees, such as the amount of CO₂ fixed by the above-ground biomass (Wu *et al.*, 2012). There are many factors which have an effect on the total store of C within the orchard system, such as fertilizer inputs and microbial respiration. The C stored within leaf material is either translocated into the woody parts of the tree or fall from the tree and decompose into soil organic matter (SOM), while the C within the apple fruit is usually removed from the system during the harvesting process (Wu *et al.*, 2012).

2.1.5 Aims and objectives

Although various estimates of global C pools exist, there is limited knowledge about different systems and their C storage as the calculations are complicated by a number of factors. These factors include the diversity of soil types that exist, the paucity of information for each soil type and how they are affected by the changing climate and the spatial variation in C content (Batjes, 1996). The soil C levels will also differ due to physical characteristics of the soil type, for example, the parent rock below the soil and how that weathers, and different land uses and management practices (Batjes, 1996). The main project aim is to determine the C storage potential of orchards in relation to the management system as management practice has previously been shown to have an effect on the soil C store in New Zealand orchards (Deurer and Sivakumaran, 2008). This will ultimately lead to the estimation of the environmental value of C stored and add to the global knowledge of C sequestration by understanding the levels of C within the orchard system. The focus of the study is to determine the C stored in above and below-ground biomass and soils in different orchard management types. The main management types are those of the traditional method of planting (usually found in orchards with trees > 15 years old) and the method of growing orchard trees along a trellis network (< 15 year old trees).

2.2 Materials and methods

2.2.1 Site description

Between March and September 2009, three cider orchards and six commercial apple orchards were sampled for the initial C survey (Table 2.2). They were chosen to provide a range of apple variety, stand age, soil type and management practices (Table 2.3). Many commercial orchards have adopted more intensive planting regimes, resulting in smaller trees, thus cider orchards were included to give a larger range of tree size. As a result of less intensive management, traditional orchards are subjected to much less soil disturbance than intensively managed orchards, thus providing important data for comparison. All of the orchards in site D that were visited in the initial C survey (top 5 cm of soil) of 2009 were revisited in January 2011 to collect soil samples down to 1 m depth.

Table 2.2. Site descriptions including location, soil texture and type of orchard sampled for each orchard sampled for initial carbon survey in 2009 (n = orchards at that site). The traditional cider orchards contained a number of different apple varieties.

Site	Location	Soil texture	Orchards sampled	n
A	52°3' N 2°38' W	Silty clay	Traditional cider	1
B	52°9' N 2°41' W	Silty clay loam	Traditional cider	1
C	52°4' N 2°52' W	Silty clay	Traditional cider	1
D	51°5' N 53°18' W	Upper greensand	Cox	3
			Bramley	3
E	51°11' N 31°3' E	Clay loam	Cox	1
			Bramley	1
F	51°14' N 56°8' E	Loam	Cox	3
			Bramley	3
G	52°6' N 2°32' W	Silty clay loam	Cox	1
			Bramley	1
H	51°10' N 45°52' E	Loam	Cox	1
			Bramley	2
I	52°41' N 4°43' E	Heavy silt	Cox	1
			Bramley	3

Table 2.3. Variety, stand age (years), orchard size (ha) and stand density (stems per ha⁻¹) data for the orchard fields visited in the initial C survey, 2009. Site B has a stand age ranging from 10 -100 years due to older trees being replaced with younger trees as and when they die and are removed from the orchard. The ground flora for all of the sites were grass strips between the tree rows, for sites A – C, these were grazed, however sites D – I were mown and not grazed.

Site	Variety	Stand age (years)	Orchard size (ha)	Stand density (stems per ha ⁻¹)
A	Traditional cider	60	10.10	246
B	Traditional cider	10-100	5.40	549
C	Traditional cider	12	1.80	46
D.1	Cox	5	1.84	1016
D.2	Bramley	6	3.11	618
D.3	Bramley	4	2.00	1108
D.4	Bramley	28	2.00	484
D.5	Cox	22	3.52	569
D.6	Cox	1	2.75	2850
E.1	Cox	2	0.58	2793
E.2	Bramley	2	0.13	2308
F.1	Cox	25	4.87	1484
F.2	Bramley	20	1.95	856
F.3	Bramley	27	1.22	501
F.4	Bramley	19	1.87	501
F.5	Cox	4	2.21	1905
F.6	Cox	27	1.25	501
G.1	Cox	22	2.00	1487
G.2	Bramley	22	1.00	1486
H.1	Cox	17	6.85	1098
H.2	Bramley	26	1.39	604
H.3	Bramley	26	1.43	1103
I.1	Cox	12	1.60	840
I.2	Bramley	15	1.50	839
I.3	Bramley	20	2.0	840
I.4	Bramley	30	5.0	247

2.2.2 Determination of above-ground biomass

Above ground C storage of trees was determined by measuring the height of trees (m) and the diameter at breast height (dbh) (cm) within each orchard sampled. Site A was the first site visited and 115 trees were measured, the decision was then made that for all the other sites, n = 40 over the whole orchard would give a representative sample number and be more time efficient. A hand-held Suunto clinometer was used to calculate height based on trigonometry (Van Laar and Akça, 2007). Dbh was determined using diameter tape, which is graduated with a linear scale to give tree circumference and the corresponding diameter (Van Laar and Akça, 2007). The data were analysed using the *Malus* specific allometric equation derived by Johnson and Gerhold (2001):

$$C \text{ (kg)} = 0.0217 \times (\text{dbh}^2 \times \text{height})^{1.1574}$$

The curvilinear equation was determined by Johnson and Gerhold (2001) using typically medium to large sized *Malus* species (n = 10). The trees were planted as linear stock with a dbh range from 2.89 to 11.68 cm and a height range of 3.11 to 7.53 m. This curvilinear equation accounted for 74% of the variation (Johnson and Gerhold, 2001).

2.2.3 Determination of below-ground biomass

Johnson and Gerhold (2003) used whole tree sampling in order to determine an accurate equation in which to determine the amount of C stored in the orchard tree roots. It was found that the dbh (cm) data can be used in the following regression formula to determine C content for the root system:

$$C \text{ (kg)} = 0.05836 \text{ (dbh}^2\text{)}$$

(Johnson and Gerhold, 2003).

This regression formula was determined based on data from a range of *Amelanchier*, *Malus*, *Pyrus* and *Syringa* trees (n = 46) and accounted for 97% of the variation (Johnson and Gerhold, 2003).

2.2.4 Collection of soil samples

2.2.4.1 Initial C survey

Ten soil samples were collected per hectare of each orchard field from sites A-I, with a 30 m strip around each field being excluded to avoid any edge effects. The vegetation was removed and soil down to a depth of 5 cm (volume 100 cm³) was collected and stored at 4°C. Although plant residues (litter) and fruit fall to the ground become incorporated into the soil and add to the C storage, it was not considered in the calculations of this study.

2.2.4.2 Soil C study at depth

Each of the orchards (n = 6) that were sampled during the initial C survey of Site D were revisited and 6 cores down to 1 m depth, where possible, were collected from across the orchard. A 30 m strip along the outer edge of the orchard was excluded in order to avoid any edge effects. The location of each core was chosen based on the C data

obtained from the initial C survey and the soil C maps that were created using the data obtained during the 2009 study. An overall view of the orchard C levels was desired, so two cores were taken from areas of low C, two were taken from areas of high C (determined from the initial orchard survey of the top soil, 5 cm) and the remaining two were taken from areas approximately between the two in terms of C levels.

2.2.4.3 Surrounding land use data collection

During the initial C survey throughout 2009, soil samples were taken from surrounding land uses in order to draw comparisons between the C storage of the orchards and these surrounding land uses. Sample sites were identified around the orchard fields, and sampled where there was access, as not all of the fields surrounding the orchards belonged to the orchard owners that had consented to the sampling on their land. The edge 30 m of each field were excluded to avoid any edge effects, soil samples (n = 10) were taken in a “W” shape across the whole field at equal distances. At each sample point, the vegetation and top soil were removed and soil to the depth of 5 cm and volume 100 cm³ was collected and stored at 4°C.

2.2.5 Soil laboratory analysis

2.2.5.1 pH and EC

Soil pH and electrical conductivity (EC) was measured for each soil sample collected during the initial survey in 2009 and the follow up work in 2011 using distilled water to make a soil suspension ratio of 1:2.5 (w/v). pH was then measured using a pH meter (Hanna Instruments pH 209) and EC measurements were taken using an EC meter (Hanna Instruments EC 215).

2.2.5.2 Loss-on-ignition (LOI) C analysis

The soil was dried at 105°C for 24 hours, ground and sieved (< 2 mm). Soil moisture and stone-corrected bulk density were calculated and C content was determined through LOI. The following was used to determine stone-corrected bulk density:

$$\text{Stone corrected bulk density} = \frac{\text{oven dry weight} - \text{weight stone fraction}}{\text{core volume} - \text{stone volume}}$$

Equivalent volumes (approximately 10 g) of oven-dried soil were placed into crucibles, weighed, then combusted in a muffle furnace at 450°C for 16 hours (Ball, 1964). Samples were then cooled in a desiccator and weighed. LOI is the measurement weight loss of a soil sample when using a moderate to high temperature to oxidise SOM. The weight loss is proportional to the amount of SOM contained within a sample (Konen *et al.*, 2002). LOI was calculated using the following equation:

$$\text{LOI (\% SOM)} = \left(\frac{\text{oven dry soil weight} - \text{soil weight after combustion}}{\text{oven dry soil weight}} \right) \times 100$$

The % SOM was then converted to % C analysis by the multiplication of % SOM by 0.44. These values were then converted into tonnes C ha⁻¹. All soil values were corrected using the stone corrected bulk density in order to determine the amount of C present by volume of the sample (Batjes, 1996). The inorganic C (as carbonate) was not calculated in this study.

2.2.5.3 Leco C analysis

While LOI measurements are cost effective, simple to run, allow the analysis of large numbers of samples and use small amounts of equipment (crucibles, drying oven, muffle furnace and balance), the method may have limitations and it has been found to overestimate SOC levels in some soil samples (Konen *et al.*, 2002; Yerokun *et al.*, 2007). The dry combustion method of analysis, where organic C is oxidised to and measured as CO₂, is recognised as the most accurate measure of C content (Konen *et al.*, 2002). However, as dry combustion analysis is expensive, a selection of the soil samples were chosen (a range of low through to high C content from the LOI method, n = 42 and 3 replicates of each) and analysed on the Leco CN analyser for % C in order to produce a correlation curve for C content.

Samples were weighed out to approximately between 1 and 2 g and wrapped in aluminium cups for the dry combustion analysis on the Leco analyser (Leco Instruments, Truspec CN analyser). Total % C values were calculated. A calibration curve was then determined (Figure 2.1) from which the following regression equation was derived:

$$y = 1.2826x - 1.2626$$

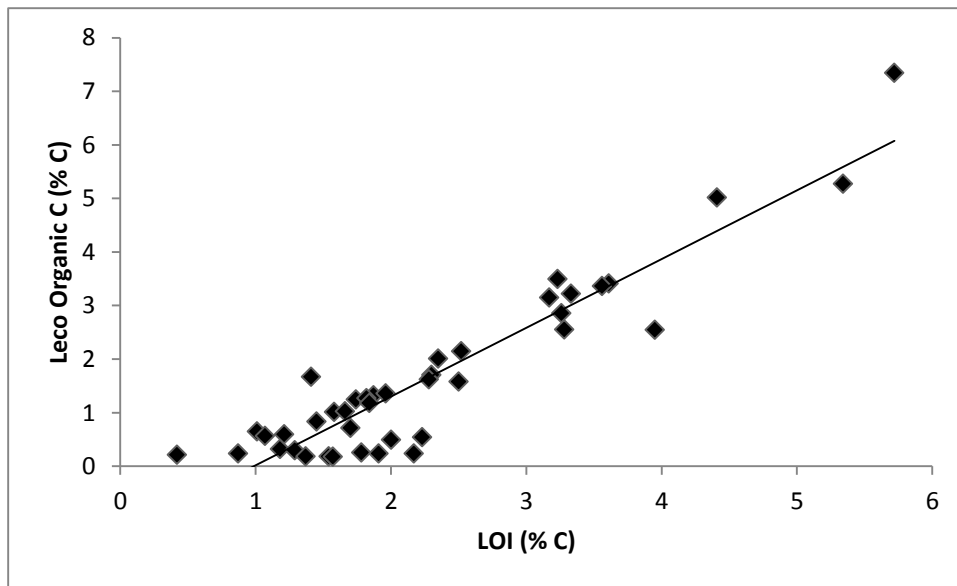


Figure 2.1. Loss-on-ignition-Leco relationship for UK orchard soils ($r^2 = 0.87$).

All the LOI C results were then corrected using the regression equation derived from the LOI-Leco correlation relationship. There were two samples in site A that were removed due to becoming negative after the correction factor was applied.

2.2.6 Data analysis

2.2.6.1 Initial C survey

C stored in soil and woody biomass of UK orchards was determined as a function of orchard age (which gives an indication of management regime as trees > 15 years are the traditional method of planting and trees < 15 years are planted along a trellis network) and apple variety (cider, Cox and Bramley apples). The analysis was carried out on soil data collected and corrected with the regression equation derived from the LOI-Leco correlation relationship (Section 2.2.5.3). All data were tested for normality using a one-sample Kolmogorov-Smirnov test and equal variances (Levene statistic) using SPSS version 20. Statistical procedures were carried out using the statistical package SPSS PC version 20 (SPSS Inc. Chicago, USA), with $p = 0.05$ used as the upper limit for statistical significance.

2.2.6.1.1 Soil C

Analysis was carried out on the soil C data with age to give a broad overview of how the age (and subsequently, planting regime) affected the level of C stored. The data were

not normal ($p = 0.001$) even after transformation (\log_{10} and $\sqrt{}$) but there was equal variance (Levene statistic $p = 0.085$), therefore a one-way ANOVA and post-hoc Bonferroni test (due to there being an uneven sample distribution) were completed. The data were further analysed using a multi-factorial test to determine the effect of age class and apple variety for the commercial orchards sampled (cider orchards were excluded from this test as there were much fewer sites sampled resulting in unequal distribution of variance). This dataset was normal ($p = 0.173$) and displayed equal variances (Levene statistic $p = 0.203$), thus a univariate ANOVA was carried out.

2.2.6.1.2 Above-ground biomass C

Data for the orchard trees were not normal ($p < 0.001$), but upon transformation to \log_{10} , the data did have a normal distribution ($p = 0.088$). The data did not have equal variance (Levene statistic $p < 0.001$), therefore ANOVA is not a valid statistical test for this data. A Kruskal-Wallis test was carried out to determine any differences between apple variety and another Kruskal-Wallis test was carried out to determine any differences between the age classes (< 15 years, > 15 years and 35+).

2.2.6.1.3 Below-ground biomass C

Data for the below-ground biomass were not normal ($p < 0.001$) even upon \log_{10} and $\sqrt{}$ transformation ($p < 0.001$). The data did not have equal variability (Levene statistic $p < 0.001$), thus ANOVA was not statistically valid. The non-parametric Kruskal-Wallis test was performed to determine any differences between the orchard variety and another Kruskal-Wallis was carried out to determine any differences between the age classes.

2.2.6.1.4 Soil C maps

Maps of each field sampled were created (Appendix 2.1) using the C data from the soil inventory as a visualisation of the C distribution within each field throughout the sites. The data were interpolated using the inverse distance weighted (IDW) method (ArcGIS version 9.3.1 ESRI, 2009). This deterministic interpolation method assigns values to locations based on the surrounding measured values and is a function of distance (ESRI, 2011). All maps were produced using ArcMap (ArcGIS version 9.3.1 ESRI, 2009).

2.2.6.2 Soil profile C

Soil moisture, stone corrected bulk density and C content was determined on the samples taken from the soil profile. A mean was taken of the soil C (t ha^{-1}) at each depth measured (0-5, 5-10, 10-20, 20-30, 30-40, 40-50, 50-60, 60-70, 70-80, 80-90 and 90-100 cm) to give an indication of the amount of C stored at each depth. A sum was then calculated to give a value of C storage throughout the soil profile.

2.2.6.3 Surrounding land use data

Surrounding land use data were tested for normality using a one-sample Kolmogorov-Smirnov test and equal variances using SPSS version 20. Data were not normal ($p = 0.006$) and did not have equal variances (Levene statistic $p < 0.001$). Data were transformed to \log_{10} data and sqrt data but were still not normal ($p < 0.001$, $p = 0.004$ respectively). As the assumptions cannot be met for a one-way ANOVA due to the high amount of variability in the data, a Welch's one-way ANOVA was performed with a Bonferroni post-hoc test to determine where any differences may lie.

2.3 Results

2.3.1 Orchard C data

2.3.1.1 Traditional orchard soil C

The soil C content for the traditional cider orchards were not normal, even when transformed to \log_{10} or sqrt data. However, the sqrt data did have equal variability (Levene statistic, $p = 0.171$) so a one-way ANOVA was carried out with a Bonferroni post-hoc test, due to the sites having unequal samples ($n = 81, 18$ and 54 for Sites A, B and C respectively). There was a significant difference between sites A and C ($p < 0.001$) and between sites B and C ($p < 0.001$). Figure 2.2 shows that sites A and B with the much older cider trees present had much higher soil C than site C which had much younger trees planted. The mean C of the sites clearly shows that A and B were not significantly different from each other (with means of 16.05 and 16.90 t ha^{-1} respectively) than they are to site C (mean C is 3.87 t ha^{-1}).

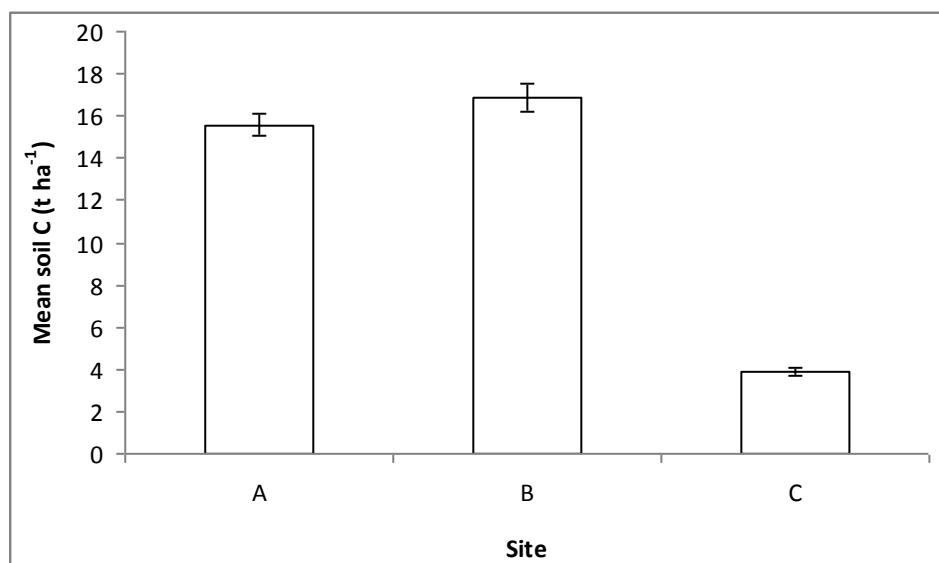


Figure 2.2. Mean soil C (t ha^{-1}) showing standard error bars from the top 5 cm layer in the soil profile found in traditional cider orchards. Site A being the oldest (<100 years old, $n = 83$), B being a mixed-aged stand (10 - 100 years old, $n = 18$) and C being 12 years old ($n = 54$).

2.3.1.2 Traditional orchard above-ground and below-ground biomass C

2.3.1.2.1 Above-ground biomass

The mean dbh for the traditional cider orchard trees was 23.1 cm (ranging from 1.8 to 58.9 cm) and the mean height was 8.84 m (ranging from 1.25 to 15.5 m). The non-parametric Kruskal-Wallis test indicated that there was a significant difference between

the sites ($p < 0.001$). The mean ranking indicates that site A had the highest amount of C in the above-ground biomass, while site C had the lowest amount of C (Table 2.4, Figure 2.3). Site A had a mean of $218.16 \text{ t C ha}^{-1}$, site B had a mean of $31.36 \text{ t C ha}^{-1}$ and site C had a mean of $25.18 \text{ t C ha}^{-1}$.

Table 2.4. DBH and height range of the traditional cider orchard trees sampled for sites A ($n = 115$), B ($n = 40$) and C ($n = 40$).

Site	DBH range (cm)	Height range (m)
A	8.9 – 57.9	6.25 – 15.5
B	2.7 – 58.9	2.0 – 12.0
C	1.8 – 13.9	1.25 – 9.5

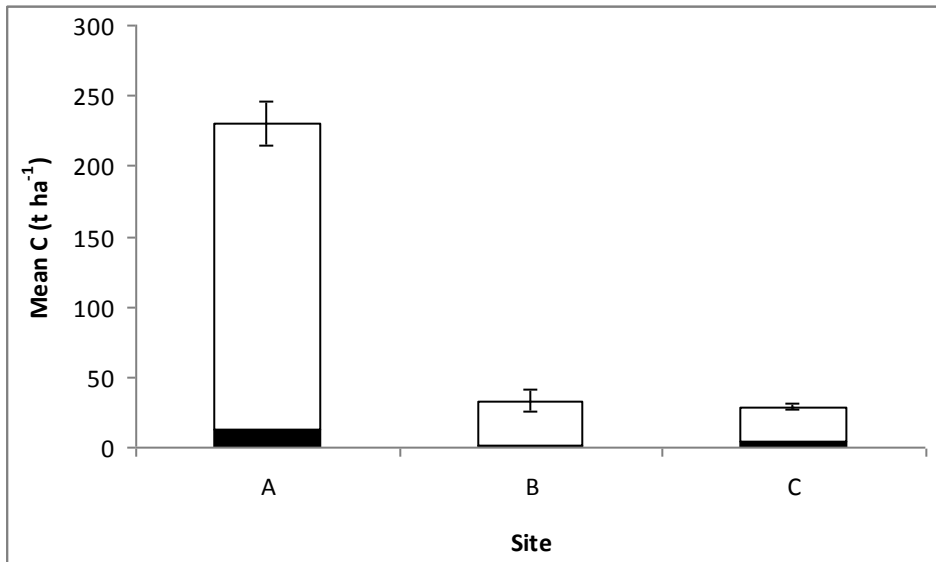


Figure 2.3. Mean C (t ha^{-1}) in above-ground biomass (white) and in the roots below-ground (black) for the traditional cider orchard sites A ($n = 115$), B ($n = 40$) and C ($n = 40$).

2.3.1.2.2 Below-ground biomass

The non-parametric Kruskal-Wallis test indicated that there was a significant difference between the sites ($p < 0.001$). The mean ranking indicates that site A had the highest amount of C in the below-ground biomass, while site C had the lowest amount of C (Figure 2.3). Site A had a mean of $12.18 \text{ t C ha}^{-1}$, site B had a mean of 1.95 t C ha^{-1} and site C had a mean of 3.57 t C ha^{-1} .

2.3.1.3 Commercial orchard soil C

The data for all of the orchard soil C (including the traditional cider orchards) were not normal ($p = 0.01, 0.000$ and 0.002 for the non-transformed data, \log_{10} and sqrt) and the data did not have equal variance ($p = 0.085$). A one-way Anova and post-hoc Bonferroni test (carried out due to an uneven number of samples) shows that there is a significant difference ($p < 0.001$) between all three age groups (< 15 years old, > 15 years old and 35+ years). The soil of orchards planted over 35 years ago had a mean C of 16.21 t ha^{-1} , the orchards > 15 years old had a mean C of 12.63 t ha^{-1} while the youngest orchards planted < 15 years ago had a mean of 7.02 t C ha^{-1} (Figure 2.4).

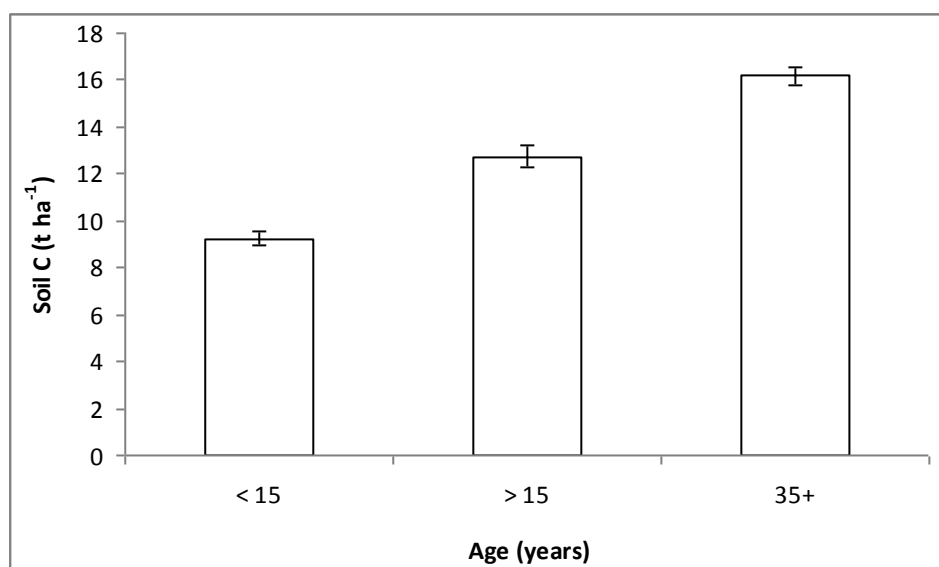


Figure 2.4. Soil C for all orchards sampled (< 15, > 15 and 35+ years old; sample depth = 5 cm) showing the mean C value and standard error ($n = 203, 216$ and 99 respectively).

Figure 2.5 indicates the mean soil C for the commercial orchards sampled in ascending age group. Although there is a general upward trend with increasing age, there is a lot of variability within the data.

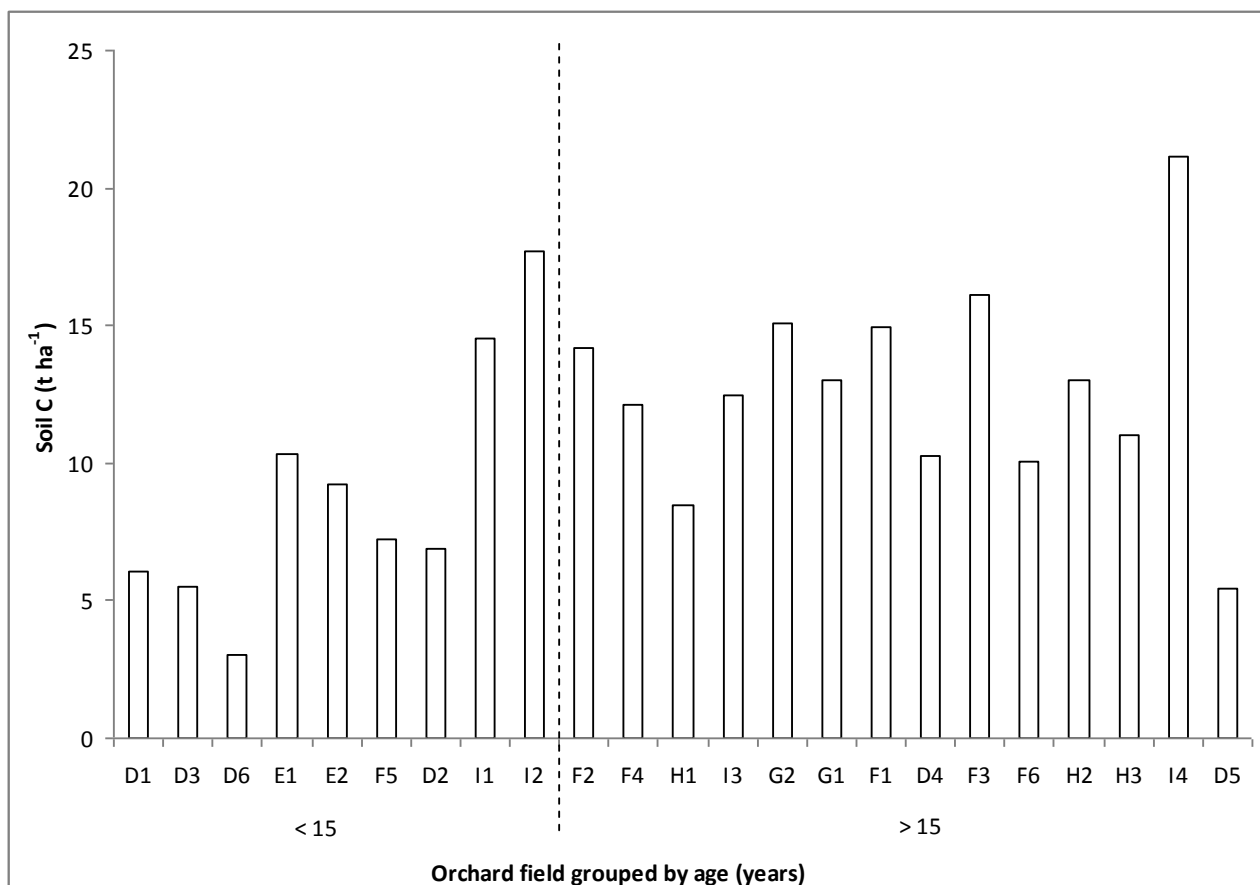


Figure 2.5. Mean amount of soil C (t ha⁻¹) in the top 5 cm calculated for the more intensively managed commercial orchards for Sites D to I, grouped by ascending age group.

The data were analysed to determine any differences between orchard age and apple variety. The data were normal ($p = 0.173$) and had equal variances (Levene statistic, $p = 0.203$). A univariate ANOVA was then performed in order to determine any interactions of age and variety. There was a significant difference ($p = 0.005$) between the variables (Figure 2.6). Bramley apple trees (mean $< 15 = 8.86$ t C ha⁻¹, mean $> 15 = 14.33$ t C ha⁻¹) had a higher level of C than Cox trees (mean $< 15 = 7.60$ t C ha⁻¹, mean $> 15 = 10.34$ t C ha⁻¹) in both age groups.

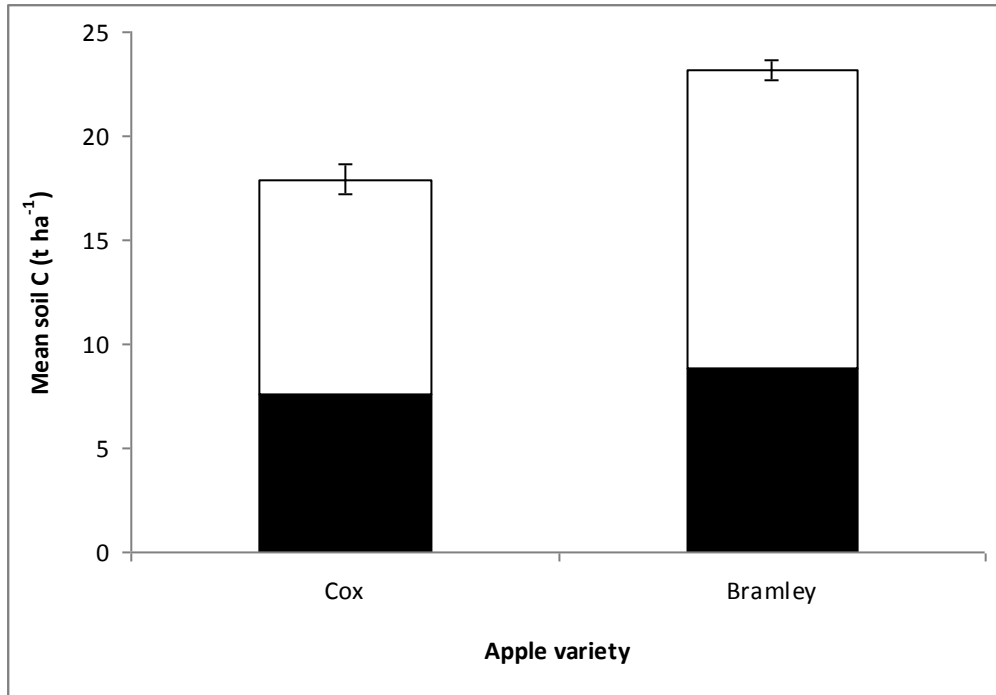


Figure 2.6. The mean C (t ha⁻¹) and standard error for the Cox and Bramley varieties sampled from the commercial orchards grouped by age; < 15 years old (black) and > 15 years old (white).

2.3.1.4 Commercial orchard above-ground and below-ground C

2.3.1.4.1 Above-ground biomass

The mean dbh for commercial orchard trees was 7.23 cm (ranging from 1.0 to 22.5 cm) and the mean height was 2.94 m (ranging from 0.75 to 6.25 m). The data for the C stored in the above-ground biomass were not normal even upon transformation using \log_{10} and sqrt methods ($p < 0.001$ in all cases) and did not have equal variances ($p < 0.001$) due to there being a high amount of variability in the data (Figure 2.7).

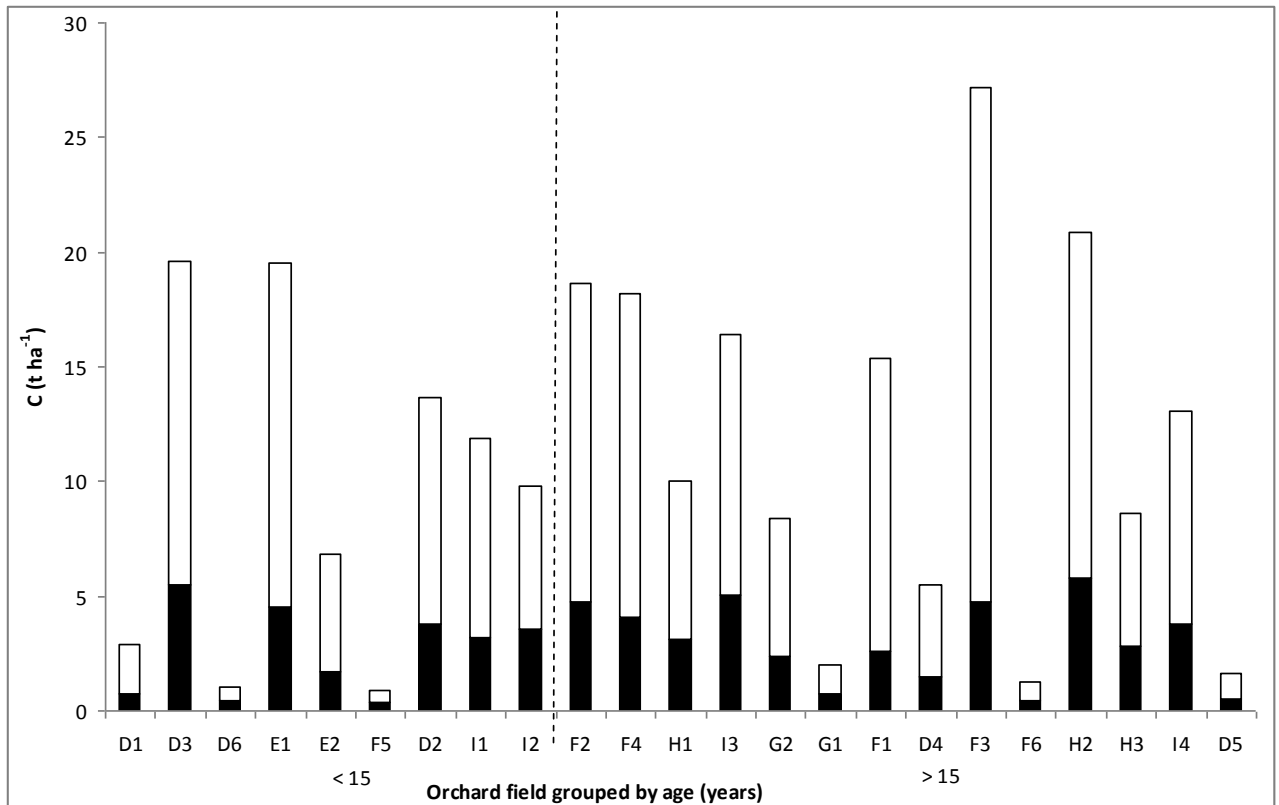


Figure 2.7. Mean C (t ha^{-1}) in above-ground biomass (white) and in the roots below-ground (black) for the more intensively managed commercial orchards for Sites D to I, grouped by ascending age group.

A Kruskal-Wallis test was carried out for both the age group of the orchard and the apple variety. There was a significant difference ($p < 0.001$) in both cases. In terms of age, commercial apple orchards < 15 years old had a mean of 6.91 t C ha^{-1} , while those > 15 years old had a mean of 8.92 t C ha^{-1} stored in the above-ground biomass. There was a significant difference between apple varieties ($p < 0.001$) with Bramley apple trees containing a mean of $10.57 \text{ t C ha}^{-1}$ and the Cox containing a mean of 4.99 t C ha^{-1} (Figure 2.8).

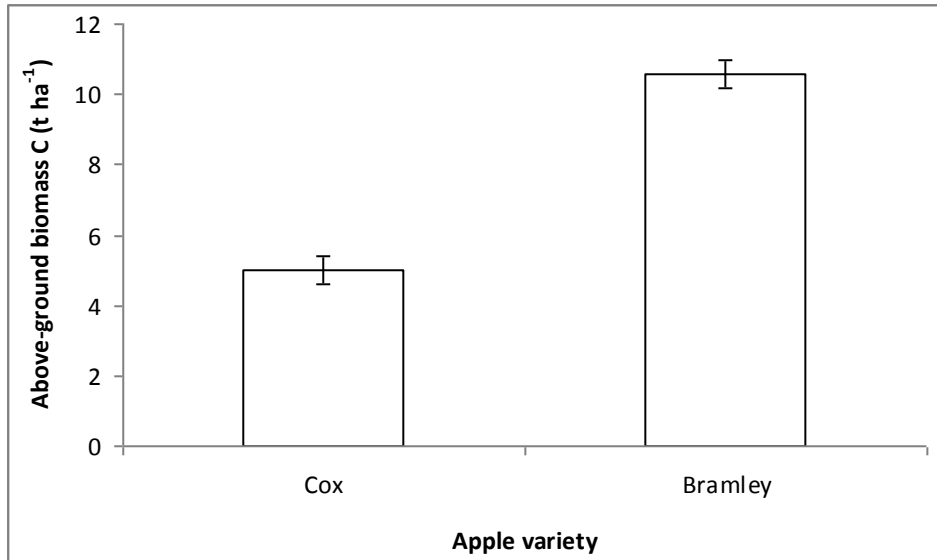


Figure 2.8. The mean values and standard error of C (t ha⁻¹) for the above-ground biomass of each apple variety (Cox, n = 400; Bramley, n = 520).

2.3.1.4.2 Below-ground biomass

Figure 2.7 shows that there is a general increasing trend with orchard age, which is confirmed by the Kruskal-Wallis test showing that there is a significant difference ($p < 0.001$) between the orchards < 15 (mean 2.66 t C ha⁻¹) and the orchards > 15 (mean 3.29 t C ha⁻¹). There was also a significantly higher ($p < 0.001$) amount of C stored in Bramley orchards than in Cox orchards (respective means 4.11 and 1.66 t C ha⁻¹).

2.3.1.5 Soil profile C

The mean C (t ha⁻¹), pH and EC are shown in Table 2.5 and it is important to consider that due to the stoniness of Site D, fewer samples were taken lower down the soil profile, thus there was a much higher variability found within the samples. This lower sample number implies that the results for the level of C are less reliable further down the profile (Batjes, 1996). The sum of the C over the soil profile is 22.91 t ha⁻¹, with 77% of the total C being held in the top 30 cm and 88% of the total C in the first 50 cm.

Table 2.5. Mean soil C (t ha^{-1}), pH and EC for the orchard soil profile for both Cox and Bramley orchards at Site D.

Depth (cm)	n	Mean soil C (t ha^{-1})	CV (%)	Mean pH	Mean EC
0 – 5	36	7.32±0.77	63	6.3±0.04	0.26±0.01
5 – 10	36	6.86±0.82	72	5.4±0.03	0.22±0.00
10 – 20	36	2.00±0.27	82	5.4±0.04	0.21±0.01
20 – 30	35	1.48±0.32	128	5.4±0.05	0.18±0.01
30 – 40	32	1.00±0.16	89	5.4±0.06	0.16±0.01
40 – 50	26	1.47±0.35	122	5.5±0.06	0.15±0.01
50 – 60	16	0.87±0.31	145	5.4±0.08	0.16±0.01
60 – 70	9	0.92±0.57	187	5.5±0.10	0.19±0.02
70 – 80	6	0.54±0.37	168	5.5±0.16	0.14±0.02
80 – 90	4	0.46±0.45	197	5.7±0.00	0.16±0.00

2.3.2 Surrounding habitat

The different land use types sampled and their pH and EC characteristics are detailed in Table 2.6.

Table 2.6. Mean soil pH and EC data (top 5 cm) for the initial C survey in 2009 for the different habitat types sampled with the standard error indicated.

Habitat	Mean pH	Mean EC
Cider orchard	5.5±0.05	0.35±0.00
Cox orchard	5.9±0.04	0.38±0.01
Bramley orchard	6.2±0.03	0.34±0.00
Perry orchard	4.7±0.04	0.30±0.01
Woodland	4.1±0.21	0.35±0.01
Arable	6.6±0.06	0.42±0.04
Hops	5.5±0.04	0.35±0.02
Grassland	6.3±0.05	0.36±0.01

Figure 2.9 shows that there is reasonable certainty (95% confidence) that the arable habitat is different from the other habitats as the error bars do not overlap with any of the other habitats. It also shows that there is 95% confidence that the means of the apple orchards (cider, Cox and Bramley) do not overlap with the means of the grassland habitat. There is difficulty in finding equal variances within the data as the orchards were the main focus of study so there is a much larger n for the orchards (153, 175, 190) than for the surrounding land use habitats (10, 13, 10, 10, 37). As it is not possible to obtain equal variances through transformation of the data, a Welch's one-way ANOVA was performed with a Bonferroni post-hoc test.

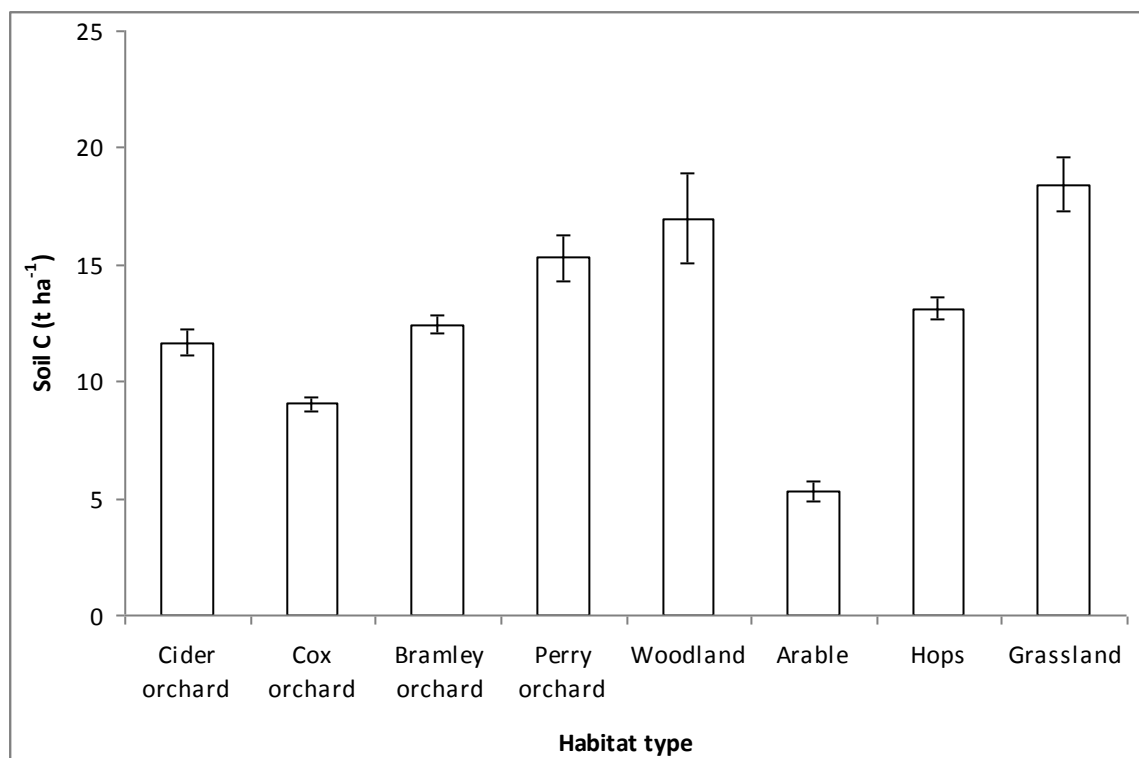


Figure 2.9. Mean soil C (t ha⁻¹) and standard error for the surrounding land use types in the initial C survey, 2009.

A nested ANOVA was carried out to determine if there was any effect of field on the data and while it gave an indication of significant differences between the fields within different habitats, due to the lack of homogeneity in the data, the results of the nested ANOVA cannot be presented. As there is not a version of a more complicated Welch's ANOVA for nested data, it was unable to be performed, therefore the broad level analysis of the habitat types by Welch's ANOVA is presented.

The Welch's one-way ANOVA showed that there is a significant difference in terms of soil C between habitat types ($p < 0.001$) with the significant differences between cider orchards and Cox orchards ($p < 0.001$), cider orchards and woodland ($p = 0.038$), cider orchards and arable fields ($p = 0.049$) and cider orchards and grassland ($p < 0.001$). There are significant differences between Cox and Bramley orchards ($p < 0.001$), Cox and perry orchards ($p = 0.015$), Cox and woodland ($p < 0.001$), Cox and grassland ($p < 0.001$). Bramley is significantly different to arable ($p = 0.015$), and significantly different to grassland ($p < 0.001$). Perry is significantly different to arable ($p = 0.007$). Woodland is significantly different to arable ($p < 0.001$). Arable is significantly different to grassland

($p < 0.001$). Hops is not significantly different to any of the other habitats ($p > 0.05$). The grassland habitat had the highest amount of C sequestered in the soil, while the arable habitat had the lowest amount of C stored. With regard to the apple orchards, Bramley had almost 50% more C stored in the soil than the Cox orchards.

2.4 Discussion

Orchard planting and management is a key determinant of the amount of C stored in the biomass. More traditional planting with larger spacing yields more standing biomass with higher levels of C stored in the soil and biomass, in contrast to the younger trees of intensively managed orchards, where biomass is kept to a minimum by planting a higher density of trees, often along trellis systems. Figure 2.10 displays the amount of C in orchards of different management types, ages and apple variety. This is in agreement with the IPCC (2000) who state that younger stands may be a source of C until C assimilation exceeds soil respiration levels and the stand then becomes a C sink. It was expected that younger fruit trees would store less C and that older trees would be a C sink, the IPCC (2000) and a study by Wu *et al.* (2012) found that apple trees below 8 years old are a C source, then when they are in the full fruit stage at 8 years old, they become a C sink. Wu *et al.* (2012) suggests that this C store peaks at around 18 years old with the capability to store C declining with age. The ability to sequester more C will be determined by any limiting factors in the stand environment, should there be few limiting factors, the trees will continue to sequester C. The rate of C sequestration may change over time.

This study found that orchards have more C stored in the soil than that of arable crops, which is in agreement with the study carried out by Milne and Brown (1997) that shows agricultural land to hold only 9.6% of the UK total amount of C (114 M tonnes). The amount in orchards, however, was considerably lower than the C stored under woodlands, which have been found to hold approximately 80% of the total amount of C stored within the UK (Milne and Brown, 1997). This could be partly due to the fact that fruit bearing trees partition a proportion of their C to the fruits. Fruit trees have been found to have a similar or even higher total dry matter than non-fruiting trees; which indicates a higher photosynthetic ability (Lenz, 2009). Orchards also consist of much smaller trees, resulting in a lower leaf area index (LAI) than trees found within a forest system, and this would result in a lower C for orchard systems (Lenz, 2009). In an orchard system, there is a focus on achieving the highest and best quality yield possible and the woody biomass is not quite so important. For woodland trees there is not a focus on fruit production and as a result the C is partitioned differently between the two

habitat types. The fixed C from the net primary production (NPP) of woodland trees is allocated 30, 43 and 27% to the leaves, wood and roots respectively, while in the orchard systems studied, it was allocated 11, 18 and 15% with 52% being attributed to the fruit (Lenz, 2009). A study carried out by Sofo *et al.* (2005) on olive and peach orchards found that young orchard trees put the majority of sequestered C into the permanent structures and root system, and as the trees mature, more C is allocated to leaves, pruning material and fruit. There is evidence from the data that in terms of soil C storage, orchards store more soil C than arable land but less than grasslands, this agrees with Bradley *et al.* (2005). Soil C can increase when croplands are converted to grasslands, plantation forest or native woodland (Ostle *et al.*, 2009). Forests and woodland contain approximately 80% of UK C stored in vegetation (Cannell, 1999; Ostle *et al.*, 2009). Agricultural systems are often regarded as C sources due to regular tillage, in contrast to ecosystems such woodland and grasslands (Panzacchi *et al.*, 2012). As the need for land for food production increases and orchard soils are found to contain significantly higher C levels than arable soils, there is potential to convert arable land into orchard. This practice has high merit in terms of C mitigation and could be eligible for land use subsidies, but would need to be balanced as there is also a demand for arable crops, they cannot all be converted into other land uses. In order to gain a full picture of the potential of orchards in terms of C storage, other apple varieties, such as those becoming more popular to consumers, should be studied to determine the C storage. As consumer interests change, the growers will change the varieties to supply the demand, while this may or may not impact the C storage potential of orchards, it is an area of investigation. Currently there is little research on the C storage of apple orchards and the positive role that orchard owners can potentially have in combating climate change (Warner, 2009). Policy makers need to have a strong scientific base on whether or not to make orchard systems eligible for C credits and to determine how to account for the C stocks within the system (IPCC, 2000). The C credits system aims to increase the rate that C is fixed and raise public awareness of reducing GHG emissions (Brown, 2007). However, it is difficult to determine exactly the amount of C within a system and then to compare it to other land use areas due to the large amount of uncertainty within the studies. Making global estimations is not always reliable due to the quality of representativeness and the reliability of data. Soil data must be kept up to

date as it is important for making predictions on the soil C store in terms of any land-use change and predicted changes in the climate (Batjes, 1996). In order to get a whole system view of orchards, the farming processes involved in the growing of apple trees must be explored in order to determine GHG emissions released and to highlight strategies for the orchard industry to reduce emissions (Warner, 2009). When system processes have been fully quantified, not only can C offset schemes be determined but also sustainable farming practices. These practices should concentrate on lowering the amount of inputs of orchards whilst maintaining apple yield and quality (Warner, 2009). This study found that the majority of the soil C is stored in the top layers of the soil profile, with 77% being held in the first 30 cm and 88% in the first 50 cm. This compares well with the findings of Batjes (1996), where it was shown that between 39 and 70% of the soil organic C (SOC) in the upper 100 cm layer is found within the first 30 cm and that between 58 and 81% is found within the upper 50 cm of the soil profile. It is reported in Batjes (1996), that there is a range of soil C in the upper 100 cm of the soil profile, from $3.1 \times 10^{-7} \text{ t ha}^{-1}$ for sandy Arenosols to $7.76 \times 10^{-6} \text{ t ha}^{-1}$ for Histosols. This study found a total of 22.91 t ha^{-1} for the orchards at Site D, which contains the soil type, Upper Greensand.

This study found that the soil had a higher mean C than the amount of C found in the vegetation. This result is in agreement with the IPCC (2000), which states that global C stocks are about 5 times higher than the C found in the vegetation of a system.

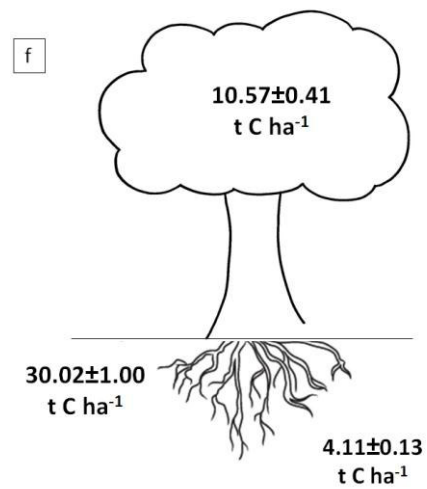
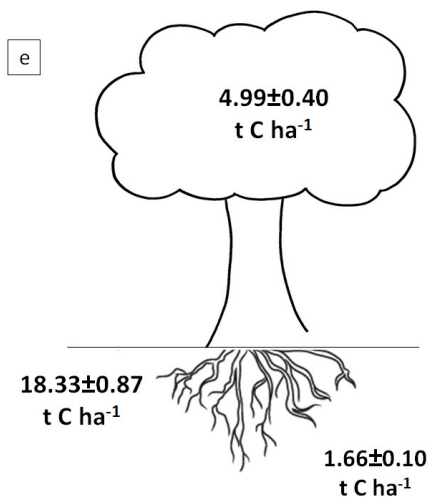
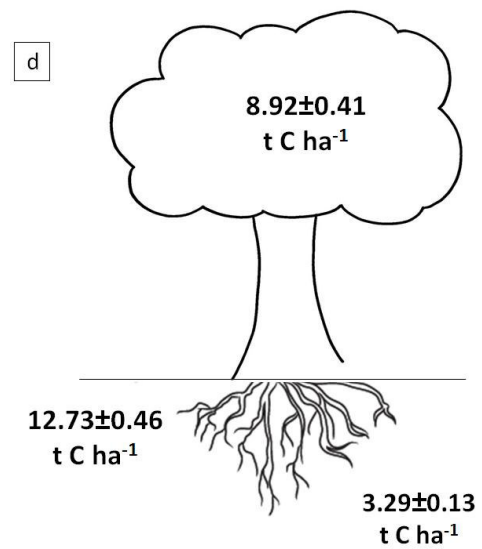
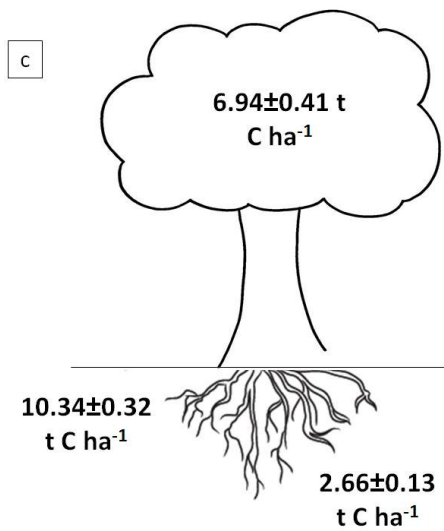
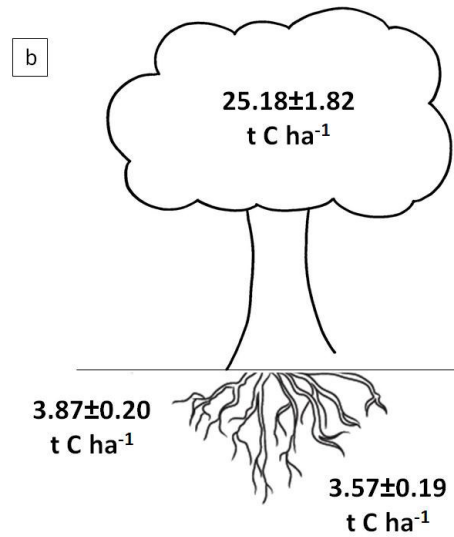
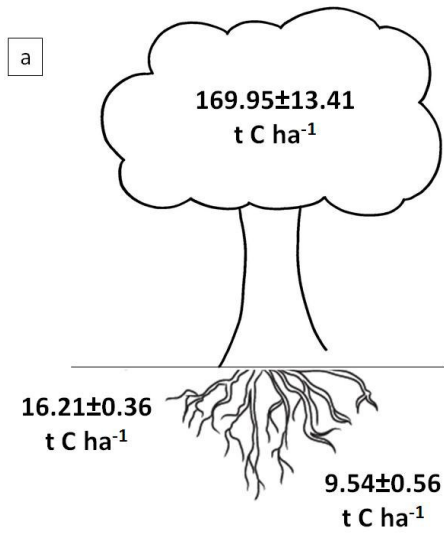


Figure 2.10. System diagrams to show the mean amount of C in UK orchard systems. Figure a. shows the C stored in traditional cider orchards (Sites A and B), Figure b. shows the C stored in bush cider orchards (Site C), while Figures c. through to Figure f. show the mean C data calculated from the commercial orchards. Figures c. and d. display the C separated by age group, with Figure c. showing the mean C for trees < 15 years old and Figure d. showing the mean C for trees > 15 years old. The soil C calculated for Figures a. to d. show the amount of C in just the top 5 cm of the orchard soil, while Figures e. to f. give the values for Cox and Bramley orchards (respectively) of the top 100 cm of the soil profile. In all Figures, the value on the bottom left indicates soil C, while the value on the bottom right indicates root C.

2.4.1 Further work

It would be useful to consider C:N ratios to indicate the level of decomposition and to determine the quality of the organic matter that is held within the soil. Batjes (1996) has reported a mean ratio ranging from 9.9 for Yermosols to 29.8 for Histosols, with a decrease in ratio with an increasing of soil depth, where there is a higher level of breakdown and increased storage of older age humus. Determining the C:N ratio of UK orchards would give greater detail of the C sequestration with the orchard system.

The allometric equations used to determine the above- and below-ground biomass (Johnson and Gerhold, 2001; 2003) were derived from *Malus* trees with the height range of 3.0 – 5.0 m and dbh range of 3.8 – 8.9 cm. As this study determines the C stored in trees with a large range of sizes (0.75 - 15.5 m height; 1.0 - 58.9 cm dbh), further work should involve whole tree sampling for a range of tree sizes to determine if the Johnson and Gerhold (2001; 2003) equations still hold for trees of very different sizes and canopy shapes. The shape of the tree differs greatly from the traditional method of planting with a large biomass and full canopy to the trellis network trees, where biomass growth is limited in order to promote maximum fruit yield. Whole tree harvesting, although not practical for this study due to cost, time and transportation limits, should be carried out in further studies to determine actual levels of C stored within the woody above-ground biomass.

In order to determine the C balance of the whole system, the litter inputs should be calculated, taking into account the residues (leaves, fruit and prunings) that fall from the tree then decompose and enter the soil pool. The majority of C partitioned to fruits leaves the system after harvest, but there is a proportion that fall to the orchard floor and the C is assimilated back into the SOC through decomposition (Lakso, 2010; Wu *et al.*, 2012). Future work should include the determination of C sequestration within apple tree roots,

potentially using C_{13} molecular tracing in order to obtain actual C stored within the root system. Studies into the soil and root respiration would enable a more holistic view on the C balance of the orchard ecosystem and determine the amount of C from the residues that is returned to the SOC or released through respiration (Wu *et al.*, 2012).

Future studies should include testing the soil profile of other soil types that orchards are planted on within the UK in order to make comparisons. The soil type used in this study for the determination of the C sequestration over the whole soil profile was Upper Greensand, a calcareous sandstone. Due to the nature of the soil, the overall amount of C for the profile was relatively low, therefore in order to get a better representation of the amount of C stored in UK orchards, the work should be repeated at a number of other soil types. Testing the samples from Site D for inorganic C found in carbonate minerals should also be carried out as it is reported by Batjes (1996) that there are large carbonate concentrations found in soil that is formed over calcareous parent material. Determining the inorganic carbonate levels alongside the organic C would give a more complete picture of the amount of C stored as the total C stored in the system would be apparent. Although Cox and Bramley varieties can be grown on any soil type due to being grafted onto a different rootstock, it is not always possible to find these two varieties on other soil types as many commercial orchards are moving away from growing them and investing on other popular varieties. As this is the case, it would be interesting to widen the varieties studied in order to get a better idea of the orchards that are being grown in the UK and how exactly the variety, soil and management type influence the C storage potential of orchards.

It is concluded that orchards sequester much larger amount of C than arable land use types and the amount of C sequestered in the soil and standing biomass increases with age. This result is in conflict with current commercial management which requires trees to be replaced every 30 years or so due to reduced production related to disease burden. The trees are grubbed up and burnt on site. In order to maximise the C sequestration of the orchard system, it would seem that the utilisation of the grubbed up biomass is key. Further work will focus on managing commercial orchards to enable greater C sequestration, however, whilst minimising impacts on productivity.

CHAPTER THREE

SOIL CARBON (C) FLUXES IN RESPONSE TO THE GRUBBING-OUT MANAGEMENT PRACTICES IN UK APPLE ORCHARDS

Abstract

In response to rising levels of atmospheric carbon dioxide (CO₂), it is important to investigate carbon (C) sinks and their stability. The soil organic C (SOC) pool is an important store of terrestrial C, containing an estimated 2500 billion tonnes to 2 m depth, making it a significant C sink. Natural exchange of C in the form of CO₂ occurs between the atmosphere and the terrestrial biosphere. However, disturbance to the ground may exacerbate this exchange, leading to a release in CO₂ from the soil. As disturbance has this effect, it is important to understand the implications of land management decisions. In the context of UK orchards, a grubbing-out process occurs once the orchard field reaches the end of its productive lifespan. The process removes the tree stumps and roots causing disturbance to the soil. The effect of this disturbance was quantified by measuring the amount of CO₂ lost from the system before, during and after the grubbing out event using an infra-red gas analyzer. The CO₂ leaving the system was subsequently partitioned to determine the effects of the management practice on the soil C store. It was found that the majority of the CO₂ (ca. 95%) leaving the orchard system was due to the abiotic release of trapped CO₂ in the soil pore spaces. It was determined that this process lasted for approximately 30 minutes after the disturbance event before soil respiration returned to a new steady state which was lower than when trees were present. The grubbing-out process liberated approximately 0.01 t C ha⁻¹ from the soil, which is a very small amount compared to the C loss from removal of standing biomass (in commercial orchards) of up to 25 t C ha⁻¹ and that held in the soil (ca. 80 t C ha⁻¹).

Key words: *anthropogenic disturbance, carbon sequestration, climate change, fruit production, land use change, soil management, soil organic carbon (SOC) pool, perturbation, tree removal*

3.1 Introduction

3.1.1 The changing global climate and the C pool

Global climate change and mitigation options are becoming more prominent due to continual rising levels of carbon dioxide (CO₂) and in response to the Kyoto Protocol. Atmospheric CO₂ concentrations have increased 31% from 280 ppm in the 1700s to 380 ppm in 2005 at a progressively faster rate and global CO₂ levels are currently at their highest concentration of the last 650,000 years (Canadell *et al.*, 2007; Lal, 2004; Raupach *et al.*, 2007). This rise is due to increasing fossil fuel combustion and land use change (Lal, 2004; West and Marland, 2002). The anthropogenic enrichment of atmospheric CO₂ has also led to an increase in land-surface precipitation in the Northern Hemisphere at a rate of 0.5-1%/decade and average global surface temperature has increased at a rate of 0.6 °C/century, which is above the critical 0.1 °C/decade rate (IPCC, 2001; Lal, 2004). These changes in climate, along with agricultural land-management and land-use change, may have a negative effect on the stability of the soil organic carbon (SOC) pool (Bell *et al.*, 2011). This pool, which comprises about 50 to 60% of the soil organic matter (SOM), is an important store of terrestrial carbon (C) thus attracting a lot of recent interest as a CO₂ abatement mechanism in the form of a soil C sink (DeLuca and Boisvenue, 2012; Lal, 2004; Rodeghiero *et al.*, 2009). Emphasis remains on reducing the amount of CO₂ emitted to the atmosphere from the burning of fossil fuels, however, a potential mitigation option exists to sequester a portion of the atmospheric CO₂ into the terrestrial biosphere (West and Marland, 2002). It is important to be aware of the C exchange between atmosphere, land and marine environments, as a proportion of CO₂ emitted from fossil fuel burning and terrestrial processes is absorbed by the terrestrial biosphere and oceans (Schimel *et al.*, 2001). Approximately half of the CO₂ released into the atmosphere from burning fossil fuels is absorbed by terrestrial and marine environments (Schimel *et al.*, 2001). Within the terrestrial environment, soil and biota C pools are of great importance to the global C cycle (Figure 1.1), with emphasis being on the SOC pool, which is estimated to contain 2500 billion tonnes to 2 m depth (Lal, 2011).

The amount of organic C found in the soil is a balance between the C inputs and outputs and subsequently, reducing CO₂ soil emissions and increasing the soil C stored is of importance

as an offset to implications of the changing climate (Gregorich *et al.*, 1998). As one of the four main global C pools, the soil C pool plays a major role in climate change mitigation options as it is four times greater than the biotic C pool, with below-ground allocation of C to roots and SOM being significantly higher in C than the above-ground woody biomass (Lal, 2004; Schlesinger, 2000; Scurlock and Hall, 1998). More than twice as much C is retained in soils than in vegetation or the atmosphere (Bellamy *et al.*, 2005). Natural sinks, such as the soil fraction, remove part of the anthropogenic CO₂ from the atmosphere and sequester it (Canadell *et al.*, 2007). However, organic C can be lost from the soil sink through the mineralization of SOM to CO₂ and minor losses can occur by soluble organic C leaching out of the system (Ball *et al.*, 1999; Gregorich *et al.*, 1998). Bellamy *et al.* (2005), found that relative to the existing soil C content across England and Wales, during the study period 1978-2003, C was lost from soils at a mean rate of 0.6% year⁻¹ irrespective of land use. The historic depletion of SOC has contributed 78 ± 12 Pg of C into the atmosphere (Lal, 2004). As any changes in the soil C content can have a large effect on the global C budget, it is important to determine and quantify any changes and possible subsequent effects they may have on the environment (Bellamy *et al.*, 2005).

3.1.2 Disturbance effects on soil C

It is known that anthropogenic perturbations to the soil cause a decline in organic matter and have the potential to dilute the effect on soil C because the disturbance mixes subsoil with relatively low organic matter, with surface soil of high organic matter content (Gregorich *et al.*, 1998). Disturbance increases the loss of CO₂ through soil respiration or by the decomposition of SOM (Schlesinger, 2000; Schlesinger and Andrews, 2000). There is a short-term CO₂ efflux to the atmosphere following cultivation practices involving disturbance to the soil, partially to do with the degassing of dissolved CO₂ from the soil solution (Calderón and Jackson, 2002). Land management activities including deforestation, biomass burning, shifting cultivation, tillage and ploughing are known to disturb soil structure and enhance the mineralisation of SOC, thereby increasing CO₂ emissions into the atmosphere (Ball *et al.*, 1999; Calderón and Jackson, 2002; Lal, 2004; Lal, 2011). Currently estimated at approximately 75 Pg C yr⁻¹ this flux is expected to increase due to soil disturbance activities (Schlesinger and Andrews, 2000). The disturbance to the ground brings crop residues into more favourable decomposition conditions with increased soil

aeration, moisture content and an increased contact with microbes, leading to greater rates of soil respiration (Gregorich *et al.*, 1998; Schlesinger and Andrews, 2000). Global losses of C from soil due to cultivation may be as large as 0.8 Pg C yr⁻¹ and as CO₂ efflux via soil respiration is recognised as one of the largest fluxes in the global C cycle, small changes could have a large effect on atmospheric CO₂ concentrations (Schlesinger and Andrews, 2000). There has been a loss of one-half to two-thirds of the original SOC from some cultivated soils (Lal, 2004). Therefore, land use change is a potential cause of CO₂ being returned back into the atmosphere, with more C being lost from soil stores through this manner and soil cultivation than from the combustion of fossil fuels up until the 1950s (Lal, 2004).

There have been few integrated studies on the quantities of CO₂ emissions following tillage practices in the UK (Ball *et al.*, 1999). Ball *et al.* (1999) carried out a greenhouse gas (GHG) flux study on a Cambisol and a Gleysol soil under spring barley in Scotland with a cool moist climate using closed chamber automatic gas sampling methods. It was determined that reduced or no tillage systems had fewer CO₂ emissions than conventional tillage, with emissions peaking between 0.14-0.15 g C m⁻² h⁻¹ for sites with no tillage, peaks of 0.225 g C m⁻² h⁻¹ for sites ploughed to a depth of 200 mm and peaks of 0.36 g C m⁻² h⁻¹ for sites ploughed to a depth of 300 mm (Ball *et al.*, 1999). Measurements on southeastern USA conventional tillage plots during four seasons (Summer, Autumn, Spring and Summer) in 2003 and 2004 measured soil CO₂ flux to be approximately 22-23 g CO₂ m⁻² h⁻¹, 4 g CO₂ m⁻² h⁻¹, 11 g CO₂ m⁻² h⁻¹ and 18 g CO₂ m⁻² h⁻¹, with variation being due to the time of year (Bauer *et al.*, 2006). Temperature and depth of cultivation are major factors in the amount of CO₂ released during tillage practices (Ball *et al.*, 1999; Bauer *et al.*, 2006). The understanding of land use and soil management of terrestrial ecosystems is therefore vital in terms of offsetting anthropogenic CO₂ emissions and for the global C budget (Lal, 2004). Improved management practices and land use change on European soils has the potential to be a net sink for 0.8% of the world's fossil fuel combustion CO₂ emission (Schlesinger and Andrews, 2000).

3.1.3 Grubbing-out management practices within UK orchards

Disturbance to the ground is observed within UK orchards as it is common practice to grub-out whole orchard fields when they become less productive. Grubbing-out is the method where the roots and stumps are cleared and the field is left fallow for up to four years before replanting (Figures 3.1 and 3.2).



Figure 3.1. Orchard tree being grubbed-out, where the whole tree including roots and stump are removed from the ground.



Figure 3.2. Orchard tree being grubbed-out, where the whole tree including roots and stump are removed from the ground.

In commercial orchards, this takes place when trees are between 15 and 35 years old, depending on the productivity of the orchard. The implications of this practice on UK orchards and their capacity for C storage are twofold. Firstly, there is the removal of the C stored within the woody biomass and secondly, the release of CO₂ from soil C stores due to the disturbance. The removal of crop residues, will leave soil unprotected, which even for short periods of time increases the risk of accelerated erosion, depletion of the SOC pool, disruption in nutrient cycling, decline in soil fauna and flora activity and species diversity, decline in water retention capacity and jeopardises the sustainable use of soil resources (Lal, 2008). The grubbing-out of apple trees may result in similar deleterious effects within the orchard system. Land-use C emissions are generated from three sources: the use of machinery; production and application of fertilisers and pesticides; and the oxidisation of SOC following disturbance (West and Marland, 2002). The extent of the disturbance is determined by the management practice; such as conventional tillage, reduced tillage and conservation tillage, where less than 15%, between 15-30% and greater than 30% crop residues are left behind respectively (Bauer *et al.*, 2006; West and Marland, 2002). Understanding the effects of the grubbing-out process on the SOC store is crucial to developing management systems to enhance soil C sequestration.

3.1.4 Soil respiration

Soil respiration (R_S) is the efflux of CO_2 from the earth's surface (Levy-Varon *et al.*, 2012). Understanding the uptake and loss of CO_2 is vital for understanding the C balance of the forest ecosystem (Högberg *et al.*, 2001a; Bhupinderpal-Singh, 2003). This is the balance between C fixed during photosynthesis and the C lost during respiration (Högberg *et al.*, 2001a; Bhupinderpal-Singh, 2003). It is fundamental that this system is understood as soil has a large C store and critical role in long-term C storage, which may be vulnerable under changing climatic conditions and large scale disturbance (Högberg *et al.*, 2005; Subke *et al.*, 2006). As the soil is a potential CO_2 sink, it is therefore important to determine if the grubbing-out practice in UK orchards has an effect on the SOC pool by monitoring the CO_2 released and the partitioning of that CO_2 loss.

R_S , which is the total CO_2 leaving the system, is derived from the sum of an autotrophic respiration component (R_A) and heterotrophic respiration component (R_H) (Bond-Lamberty *et al.*, 2004a; Boone *et al.*, 1998; Epron *et al.*, 1999; Hanson *et al.*, 2000; Högberg *et al.*, 2005; 2009; Lukac and Godbold, 2011; Subke *et al.*, 2006). R_A is the loss of CO_2 from plant roots, their mycorrhizal symbionts, rhizosphere microbiota and includes the leaching of labile C compounds from the ectomycorrhizal roots (Högberg *et al.*, 2005; Kutsch *et al.*, 2009; Levy-Varon *et al.*, 2012; Lukac and Godbold, 2011). In contrast, R_H is the microbial respiration resulting from decomposition of organic molecules in both above-ground and below-ground litter (Bond-Lamberty, 2004a; Högberg *et al.*, 2005; Levy-Varon *et al.*, 2012; Lukac and Godbold, 2009). The SOM is decomposed by soil micro-organisms, including bacteria, fungi, actinomycetes and protozoa, and soil macrofauna, such as macroscopic invertebrates and small mammals, resulting in the production of CO_2 and some other trace gases via the process of respiration (Kutsch *et al.*, 2009; Kuzyakov, 2006; Rodeghiero *et al.*, 2009). A diffusion gradient controls the soil CO_2 efflux, with more CO_2 leaving the soil environment if there is an increase in below-ground respiration under certain conditions, such as disturbance or elevated CO_2 (Lukac and Godbold, 2011). There is a higher concentration of CO_2 and a lower concentration of O_2 in the soil air than in the atmosphere. This difference drives the diffusion of O_2 and CO_2 between the two environments in order to reach a dynamic equilibrium (Wild, 1993). An increased level of CO_2 changes the diffusion

gradient and results in soil CO₂ efflux and changes in the chemical composition of SOM (Kutsch *et al.*, 2009).

The uptake of CO₂, between 109 and 120 Pg CO₂ per year globally, by plants during photosynthesis is the largest flux in the global C cycle (Janzen, 2004; Moyano *et al.*, 2009; Schlesinger, 1997; Zhao *et al.*, 2005). R_s, the primary path by which assimilated C leaves the soil surface as CO₂ (soil-surface CO₂ efflux) and returns to the atmosphere after being fixed by plants, is the second largest C flux (Davidson *et al.*, 2006; Högberg *et al.*, 2005; Kutsch *et al.*, 2009; Kuzyakov, 2006; Moyano *et al.*, 2009; Schlesinger and Andrews, 2000). R_s is analogous to the reverse process of photosynthesis and occurs continuously throughout day and night (Wild, 1993). It is the largest component of ecosystem respiration with estimates between 50% (Andrews and Schlesinger, 2001; Janzen, 2004; Moyano *et al.*, 2009; Ryan and Law, 2005) and 80% of C fixed by gross primary production (GPP) being returned to the atmosphere through soil CO₂ efflux (Janssens *et al.*, 2000). Globally, between 68 and 76.5 Pg CO₂ year⁻¹ is emitted via soil respiration (Bhupinderpal-Singh, 2003). The SOC pool, and any changes to it, has an integral effect on atmospheric CO₂ concentration as there is three times the amount of C stored in soils and plants with an annual flux of c. 60 Pg between the land and atmosphere (Lukac and Godbold, 2011).

Microbial activity is determined by the amount of SOM in the soil and this is reliant on a number of site-specific parameters, including soil temperature, moisture and the supply and quality of the substrate (Davidson *et al.*, 2006; Högberg *et al.*, 2009). Both the components of R_s are influenced by site conditions and climate and it is thought that they will respond differently to changing climate conditions, including elevated CO₂ and soil warming (Boone *et al.*, 1998; Dilustro *et al.*, 2005 Epron *et al.*, 2001; Lavigne *et al.*, 2004; Pendall *et al.*, 2004).

3.1.4.1 Differentiation of autotrophic and heterotrophic respiration

As soil respiration has a large impact on ecosystem C balance, it is important to determine how much of respiration is attributed to the autotrophic and heterotrophic components to accurately analyze and model soil respiration particularly in response to disturbance (Högberg *et al.*, 2001a, 2001b, 2005; Subke *et al.*, 2006). It is important to understand the effect on C sequestration and to determine the impact that disturbance may have on the C

balance of a system (Epron *et al.*, 1999; Hanson *et al.*, 2000; Ryan and Law, 2005; Subke *et al.*, 2006). However, it is very difficult to make distinctions between and quantify the individual component contribution. This is due to it being variable in time and space, for example, seasonal variation and autotrophs and heterotrophs displaying characteristics and carrying out the function of the other group to some degree (Bhupinderpal-Singh, 2003; Hanson *et al.*, 2000; Högberg *et al.*, 2005).

There is a wide range, from 10% to 90%, reported for the contribution of R_A (root) to total soil respiration (Hanson *et al.*, 2000; Thierron and Laudelout, 1996), with most suggesting that the figure is closer to between 50% and 60% (Nakane *et al.*, 1996; Hanson *et al.*, 2000; Epron *et al.*, 2001; Bond-Lamberty *et al.*, 2004b). This variability reflects partly on natural variability within and across ecosystems but it also highlights issues with the various methods used (Högberg *et al.*, 2005).

The main methods used to distinguish respiration proportions are:

- Take the system apart and measure each component in the lab in order to draw comparisons with the intact system. This is a highly undesirable method because it assumes that the heterotrophic organisms will continue to respire at a normal rate without the autotrophic component being present (Epron, 2009; Högberg *et al.*, 2005).
- Determine the amount of R_H by subtracting the root respiration from soil respiration (Epron, 2009).
- Use a physical barrier to exclude roots from the system (Högberg *et al.*, 2005). One such method is that of trenching where the roots around the perimeter of the treatment plot are severed by digging a trench deeper than the main rooting zone and regrowth prevented by an impenetrable lining being put down (Epron *et al.*, 1999; Subke *et al.*, 2006).
- Label the photosynthates with isotopes, either radioactive ^{14}C or stable ^{13}C . This method does not physically disturb the system but it is expensive and there is a time lag before the labelled C is distributed and then released due to differences in turnover rates (Högberg *et al.*, 2005).

- Girdling experiments which inhibit the flow of C to the autotrophic soil component can be used to estimate the contribution of R_A to total soil respiration (Högberg *et al.*, 2005). It is carried out by stripping the bark and phloem around the tree circumference to the depth of the current xylem, thereby stopping the transport of photosynthates to the roots and rhizosphere (Epron, 2009). Once the sugar pool in the roots is depleted, the fractional contribution of roots can be estimated (Högberg *et al.*, 2005). It is a method often used as changes in soil temperature and mechanical disturbance are limited by temporarily maintaining a canopy (Epron, 2009). Stand soil properties and micro-climate are not affected in the short term, however there may be an effect in the long term as there will be reduced water uptake in the stand as there will be a decreased leaf area, due to increasing litter fall and a reduction in foliage production, thus causing the soil to become wetter (Högberg *et al.*, 2005, 2009). Limitations include alterations to the microbial community activity as a result of the change in soil water and nutrient balance alongside the suppression of root exudates (Epron, 2009). Along with possible enhanced heterotrophic activity and $\delta^{13}\text{C}$ studies, which show that there may be a time delay for the autotrophic release of C that has been fixed by the trees (due to root respiration being sustained by carbohydrate reserves), there may be an underestimation in the autotrophic component during partitioning (Högberg *et al.*, 2005, 2009). A major problem with girdling trees is that it will ultimately result in death of the tree(s) studied (Högberg *et al.*, 2005, 2009).

The main criticism of soil partitioning techniques is the disturbance created to the ecosystem by physically separating the components (Epron, 2009; Högberg *et al.*, 2005). This is undesirable as it may have an effect on the rate of respiration. It is commonplace to use root exclusion techniques within forest ecosystems to determine the proportions of respiration (Epron, 2009).

3.1.4.2 Measurement of soil CO₂ efflux

There are various methods of determining changes in the soil organic C, including Repeated Inventory Approach, examining changes in the specific fractions of C and the flux approach (Rodeghiero *et al.*, 2009). The determination of soil CO₂ efflux is of great importance as soil

respiration is a major component in the C balance of terrestrial ecosystems and this method will indirectly determine the C flux into and out of the soil system over a certain period of time (Pumpanen *et al.*, 2009; Rodeghiero *et al.*, 2009). The net balance of all the C fluxes entering and leaving the soil system determines the total amount of organic C stored within that soil (Rodeghiero *et al.*, 2009).

Girdling and trenching methods of measuring CO₂ efflux, although they include the extra-radical mycelium (ERM) respiration, usually overestimate root respiration when further partitioning does not occur as mycorrhizal fungi are included in the autotrophic flux component (Pendall *et al.*, 2004). The presence and activity of soil biota are changed during girdling and trenching (Schulze *et al.*, 2005). According to root-exclusion experiments, in mature forests, autotrophic respiration accounts for about 50% of total soil respiration (Epron, 2009). Root-exclusion methods are not fully satisfactory in the partitioning of R_A and R_H (Epron, 2009).

3.1.5 Aims of the study

The aim is to quantify the CO₂ lost from an orchard system and to compare it with the amount of SOC that is lost from other land management practices causing soil disturbance (West and Marland, 2002).

The study aimed to address the following questions:

- Does grubbing-out apple orchards deplete SOC?
- How quickly do elevated CO₂ emissions from grubbing-out processes return to baseline levels?

In summary, the CO₂ lost from the orchard system will be quantified and compared to the amount of SOC lost from other land management practices. It will be determined whether the C lost is biotically lost from the organic pool or from abiotic degassing.

3.2 Materials and methods

3.2.1 Soil efflux measurements of grubbing-out events

3.2.1.1 Site description

Site 1 (loam; 53°3'N 4°16'W) consisted of a 4 year old cider orchard with Kingston Black cider trees. The field study was undertaken during May and June, 2010 when soil temperatures (depth = 10 cm) ranged between 10.5 - 33.8°C. Measurements were taken on two grubbed-out trees and a third tree, which was not disturbed in any way, for comparison. Site 2 (Upper Greensand; 52°5'N 53°18'W), consisted of a 22 year old Cox orchard (grubbed out trees measured, n = 22). The field study was undertaken during September and October, 2010 when soil temperatures (depth = 10 cm) ranged between 8.5 - 16.7°C. Site 3 (Upper Greensand; 52°5'N 53°18'W), consisted of a 30 year old Bramley orchard (grubbed out trees measured, n = 21). The field study was undertaken during March, 2011 when soil temperatures ranged between 1.9 - 12.3°C.

3.2.1.2 Measurement of soil CO₂ efflux

To measure the CO₂ efflux of the orchard system during the management practice of grubbing-out, soil respiration was determined at each grubbing-out event. This was measured using a closed chamber EGM 4 infra-red gas analyzer (IRGA) equipped with a SRC-1 gas chamber with an internal volume of 1964 cm³ and area exposed to the soil of 78 cm² (PP Systems, Hitchin, Herts, UK). The IRGA used a 124 s enclosure time to log the CO₂ concentration, a 15 s purge time to lower the CO₂ concentration inside the chamber to ambient levels and a 15 s equilibration time. The EGM 4 IRGA calculates soil CO₂ flux rates from the change in CO₂ levels over time, the volume of the system and the surface area of the soil tested (Janssens *et al.*, 2000). At the end of each measurement, a linear regression was computed between the soil CO₂ respiration and the CO₂ concentration within the chamber (La Scala Jr. *et al.*, 2000).

At all three sites, baseline CO₂ levels were measured, and CO₂ efflux was measured during the grubbing-out event and the CO₂ response was measured on the subsequent days following the disturbance, in order to monitor the soil respiration and its return to baseline

levels. At Site 1, baseline levels were measured on three occasions before the grubbing-out event (day 1) and an average was used. At Sites 2 and 3, flux measurements were taken before, during (day 1) and after the grubbing-out event. At Sites 2 and 3, flux measurements were taken sequentially throughout the orchard, in order to get a spread across the whole field (n = 4 at each grubbed-out tree). The outer 30 m were excluded from measurements, in order to avoid the edge effect. Four replicates at each tree were taken following north, east, south and west directions. As temperature flux can cause changes in respiration flux (Valentini *et al.*, 2000), measurements were taken at the same time of day and although every effort to ensure similar weather when measurement were taken, experiment timing was dictated to by when the orchards were being grubbed-out (Table 3.1).

Table 3.1. Data for the weather conditions and mean temperature (°C) for each site during the soil efflux measurements of grubbing-out events.

Site	Mean temp (°C)	Standard deviation	Standard error	Weather conditions
1	21.30	4.52	0.21	Dry, sunny
2	13.40	1.50	0.07	Dry, overcast with intermittant rain during day 3
3	6.42	1.66	0.08	Dry, sunny with intermittant rain during day 4

3.2.1.3 Data analysis

Within site analysis was completed using a one-way ANOVA and a post-hoc Tukey test, with tree as the factor. A random effects model was then run with the factor level of day. Statistical procedures were carried out using the statistical package SPSS PC version 20 (SPSS Inc., Chicago, USA), with p = 0.05 used as the upper limit for statistical significance.

3.2.2 Determination of time taken for CO₂ efflux to return to baseline after grubbing-out events

3.2.2.1 Site description

The experimental site consisted of a poplar (*Populus* sp.) stand from a temperate oceanic climatic region in Abergwyngregyn, Gwynedd, North Wales (53°14'N, 4°01'W). The mean annual soil temperature at a soil depth of 10 cm is 11 °C and the annual rainfall is 1250 mm. Field studies were undertaken during September, 2011 when soil temperatures ranged between 11.8 – 14.3 °C. Three poplar (*Populus* sp.) trees were removed from the soil on three consecutive days to replicate the grubbing-out process carried out within UK apple orchards. Poplar (*Populus* sp.) trees were studied and not apple (*Malus* sp.) because there were no grubbing-out events taking place in orchards at the time of experimentation. The trees, including the roots and stump, were pulled out of the ground mechanically, in the same way that trees are removed from an orchard. Once the tree, including the root ball, had been pulled out, the soil was replaced in the hole afterwards.

3.2.2.2 Measurement of soil CO₂ efflux

Flux measurements were taken before, during and after the grubbing-out event using a PP Systems EGM-4 IRGA with a SRC-1 chamber (see section 3.2.1.2 for detailed information on how the EGM-4 IRGA works, PP Systems, Hitchin, Herts, UK).

On day 1, tree 1 was grubbed-out and four measurements were taken sequentially following north, east, south and west directions at ten minute intervals two hours before the grubbing-out event and again one hour before the grubbing-out event. The tree was then mechanically removed, including its stump and roots. Flux measurements were taken continuously for 2 hours (one measurement approximately every 3 minutes) in the hole left by the tree. Flux measurements were then taken for 5 hours following the grubbing-out event with four measurements being taken per hour. On day 2, tree 2 had four measurements taken sequentially following north, east, south and west directions at ten minute intervals two hours before the grubbing event. Flux measurements were taken continuously for three hours (one measurement approximately every 3.5 minutes) after the tree was mechanically removed. Flux measurements were then taken for four hours

following tree removal with four measurements being taken every hour. On day 3, tree 3 had four measurements taken sequentially following north, east, south and west directions at ten minute intervals two hours before the grubbing-out event. Flux measurements were then taken for 2 hours (one measurement approximately every 5 minutes) after the tree was removed. Measurements were then taken for the following five hours, with four measurements being taken every hour.

3.2.2.3 Data analysis

To measure the rate at which CO₂ respiration returned to the baseline level after disturbance, a single first-order kinetic equation with asymptote was fitted to the data from the area of disturbance where

$$f = y_0 + (a \times \exp^{-kt})$$

and where f is the amount of CO₂ released from the soil, y_0 is the asymptotic value equating to the baseline amount of CO₂, a is a constant which measures the magnitude of the response to the disturbance, k is the rate-coefficient describing the rate of decline and t is time. The half-life of CO₂ loss from soil ($t_{1/2}$) can then be calculated as:

$$t_{1/2} = \ln(2)/k$$

Data analysis and graphs were created using Sigma Plot 11 (Systat Software Inc., Chicago, IL).

3.2.3 Determination of the key driver of soil CO₂ efflux

3.2.3.1 Site description

Sites 2 and 3 from section 3.2.1.1 (Upper Greensand; 52°5'N 53°18'W) were tested. A homogenised sample of approximately 500 g of soil was taken from both the Cox (Site 2) and the Bramley (Site 3) orchards.

3.2.3.2 The substrate-induced respiration method

In order to measure the maximum potential respiration response and basal response of the orchard soil, a substrate-induced respiration experiment was carried out on the soil from Sites 2 and 3. The method is based on the respiratory response of substrate-stimulated microbial populations to amendment with a source of excess C and energy. The respiratory response is then measured and it is possible to determine the proportion of released C that is attributed to microbial biomass.

3.2.3.3 Laboratory soil analysis

Soil samples (each 30 g, n = 12) were amended with a C and energy substrate. The samples (n = 6) had 5 ml of 50 mM glucose and 1 mM glutamate added to them (the volume was estimated on the amount required to take the soil samples to field capacity to allow maximum microbial respiration). Control samples (n = 6) were mixed with 5 ml of distilled water. Samples were analysed on a twelve channel Ciras-SC soil respirometer (PP Systems, Hitchin). Every fifteen minutes, CO₂-free air was drawn through the soils and the rate of CO₂ efflux was measured continuously for 1 min with an automatic data logger.

3.2.3.4 Data analysis

The maximum possible amount of CO₂ that may be respired from the microbial biomass in the soil was determined by dividing the maximum CO₂ (respired by the substrate-stimulated microbial population measured in the laboratory), by the base rate of CO₂ respired (by the non-substrate-stimulated microbial population measured in the laboratory) and multiplying by the baseline (field basal CO₂ flux) determined in section 3.2.1. The results are then depicted as dotted lines on Figures 3.4 and 3.5.

The total C lost from the orchard during the grubbing-out period was calculated by the addition of the CO₂ lost from the microbial biomass to that of the CO₂ lost from root respiration. The amount of CO₂ that was remaining is trapped CO₂ that is released as a result of the physical disturbance and the soil degassing:

Total CO₂ lost = microbial respiration + root respiration + trapped CO₂

The total amount of CO₂ lost was then converted the amount of C lost from the system by multiplying the amount of CO₂ by the molecular weight of C (12) and then dividing by the molecular weight of CO₂ (44). As the figure is per tree, the final amount of C lost was multiplied by the number of trees per hectare. The grubbing-out event is a one-off management practice that occurs approximately every thirty years, therefore the release of CO₂ from the soil is not a continual release. As the roots are removed from the ground, the release of CO₂ actually falls below the baseline after a short period, therefore the time taken

to return to baseline was used to calculate how much C was lost per hour on the day of the grubbing-out event.

3.3 Results

3.3.1 Soil efflux measurements of grubbing-out events

At Site 1 (Figure 3.3), tree 1 showed a significant difference on day 1, the grubbing-out event, to all other days ($p < 0.01$). Tree 2 showed a significant difference on day 1 to all other days ($p < 0.05$) except day 2 (the first day after the grubbing-out event), which did not reach baseline levels again until day 3. Tree 3 showed no significant difference. The day of the grubbing-out event (day 1) was significantly different to all the other days ($p < 0.01$).

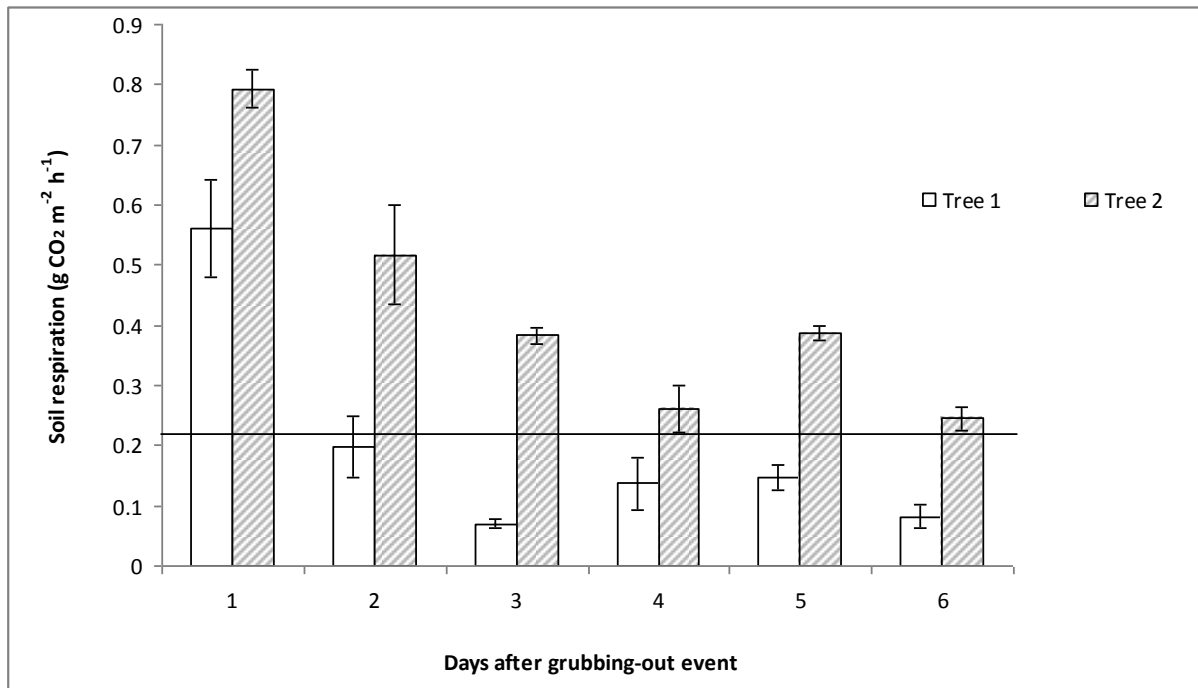


Figure 3.3. Carbon dioxide ($\text{g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$) efflux from cider orchard soil before (baseline shown as a solid horizontal line = $0.22 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$), during (day 1) and after (days 2 - 6) a grubbing-out event of orchard trees at Site 1 \pm SEM, with trees 1 and 2 being treatment trees and tree 3 the control.

Site 2 (Figure 3.4) had a significantly higher ($p < 0.05$) CO_2 efflux when trees 7 and 17 were grubbed-out than all the other trees tested in the orchard. Significant differences in mean soil respiration for the day of grubbing-out were observed ($p < 0.01$), with a mean loss of $4.43 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ per tree on the day of grubbing (maximum of $10.13 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ per tree and a minimum of $2.01 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ per tree) compared to baseline mean ($n = 22$) of $0.37 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ per tree on the previous day. The mean CO_2 lost per tree on the day following grubbing-out fell to $0.25 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ per tree. The mean amount of CO_2 released on the day of grubbing was approximately ten-fold higher than the baseline figure and the following day, the CO_2 levels fell 32.4% below the original baseline. It can therefore be estimated that $0.25 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ was attributable to microbial respiration and $0.12 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ was CO_2

released from root respiration, therefore the remaining $4.06 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ that was released on the day of grubbing was from the abiotic degassing of trapped air within the soil structure. In terms of total C lost from the system, $1.2 \text{ g C m}^{-2} \text{ h}^{-1}$, which is the equivalent to $12 \text{ kg C ha}^{-1} \text{ hr}^{-1}$, was lost but 92% of that was attributed to abiotic degassing and not a loss from the SOC pool. The total C lost as a result of microbial respiration was 0.7 kg C ha^{-1} and total lost as a result of root respiration was 0.4 kg C ha^{-1} .

No significant differences in soil respiration ($p > 0.05$) between trees grubbed-out were observed within Site 3 (Figure 3.5). The disturbance event did have a significant effect ($p < 0.01$) on the amount of CO_2 released from the soil, with a mean loss of $7.14 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ per tree (maximum of $18.08 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ per tree and a minimum of $2.31 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ per tree) compared to baseline mean ($n = 21$) of $0.18 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ per tree on the previous day. The mean CO_2 lost per tree on the day following grubbing-out fell to $0.05 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ per tree, due to the loss of root respiration. The mean amount of CO_2 released on the day of grubbing was forty times greater than the baseline figure and the following day, the CO_2 levels fell 72% below the baseline. It can therefore be estimated that $0.05 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ was lost from microbial respiration and $0.13 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ was CO_2 released from root respiration, therefore the remaining $6.96 \text{ g CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ that was released on the day of grubbing-out was from the abiotic degassing of trapped air within the soil structure. This means that in total, $1.9 \text{ g C m}^{-2} \text{ h}^{-1}$ was lost (equivalent to $19.4 \text{ kg C ha}^{-1} \text{ hr}^{-1}$) from the grubbing-out event but 98% of that is attributed to the abiotic degassing of CO_2 trapped within the soil and not a loss from the SOC pool. The total C lost as a result of microbial respiration was 0.1 kg C ha^{-1} and total lost as a result of root respiration was 0.4 kg C ha^{-1} .

Alongside, the loss of CO_2 from the soil ($0.006 \text{ t C ha}^{-1}$ for Site 2 and $0.0097 \text{ t C ha}^{-1}$ for Site 3), up to 25 t ha^{-1} C of standing biomass per tree is removed from the system by grubbing it out of the ground for an orchard field of approximately 30 years old (Chapter 2).

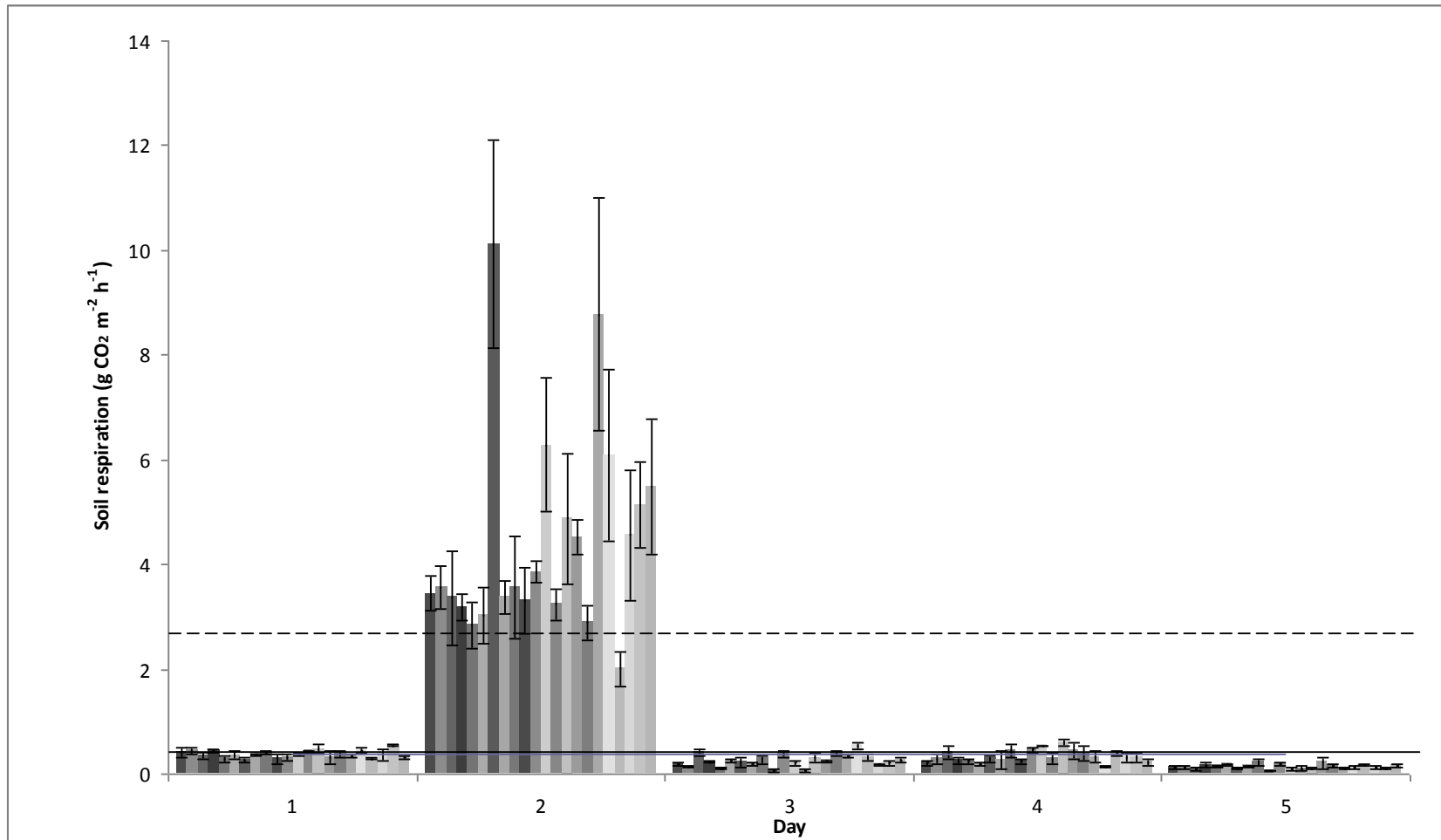


Figure 3.4. The CO₂ efflux before (day 1), during (day 2) and after (days 3-5) the grubbing-out event of a Cox orchard at Site 2, where one bar represents each tree over the 5 day measurement period. The baseline CO₂ level is represented by a solid horizontal line and was taken from the mean of the measurements taken before the grubbing-out event ($n = 22$, baseline CO₂ = 0.37 g CO₂ m⁻² h⁻¹). The dotted line represents the substrate induced respiration response of the microbes present in the orchard soil ($n = 3$, respired CO₂ = 2.65 g CO₂ m⁻² h⁻¹).

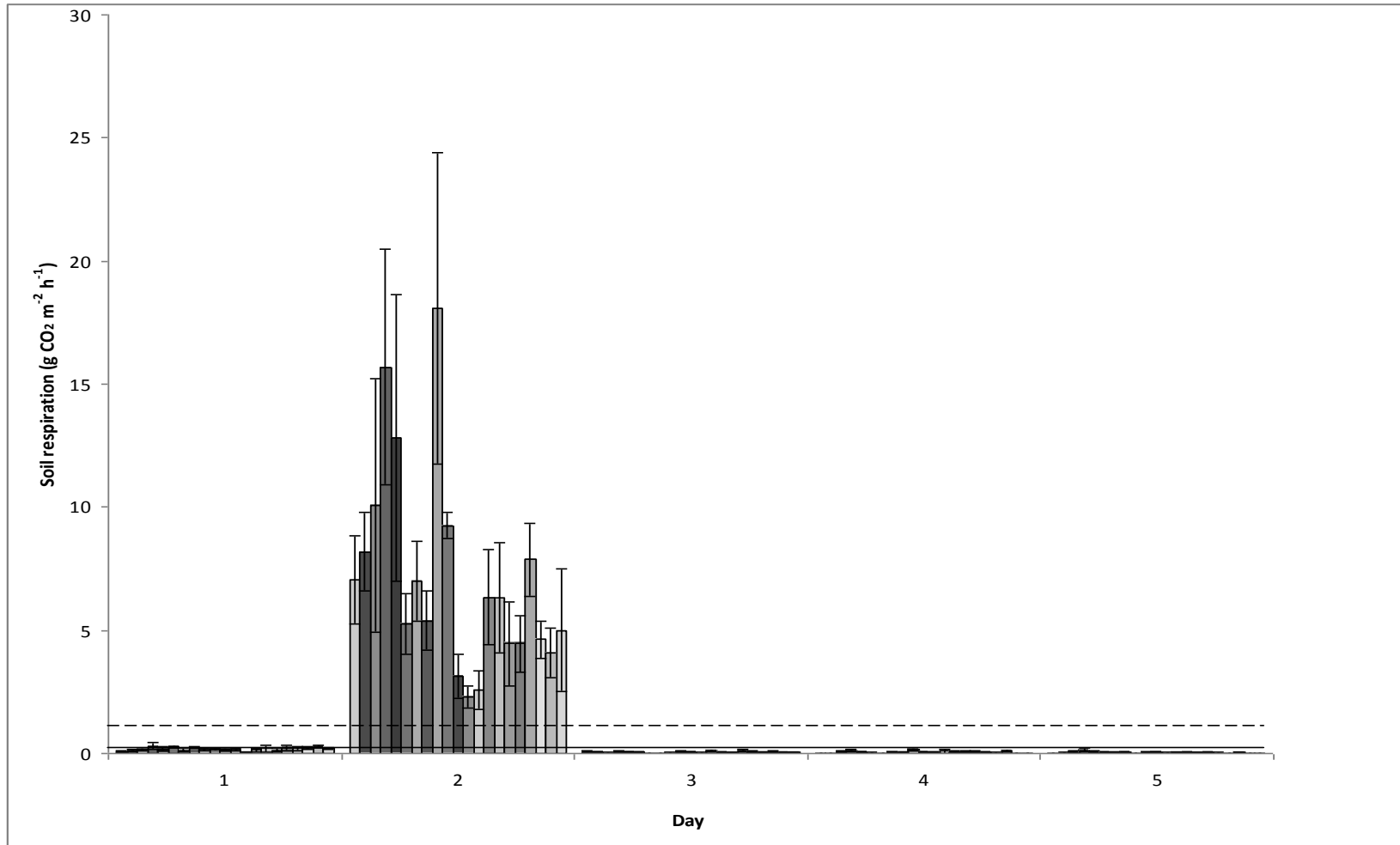


Figure 3.5. The CO₂ efflux before (day 1), during (day 2) and after (days 3-5) the grubbing-out event of a Bramley orchard at Site 3, where one bar represents each tree over the 5 day measurement period. The baseline CO₂ level is represented by a solid horizontal line and was taken from the mean of the measurements taken before the grubbing-out event ($n = 21$, baseline CO₂ = 0.18 g CO₂ m⁻² h⁻¹). The dotted line represents the substrate induced respiration response of the microbes present in the orchard soil ($n = 3$, respired CO₂ = 1.07 g CO₂ m⁻² h⁻¹).

3.3.2 Determination of time taken for CO₂ efflux to return to baseline after grubbing-out events

It was found that soil respiration rapidly decreased after tree (*Populus* sp.) removal with the levels of CO₂ released returning to baseline. Model parameters are reported in Table 3.2.

For tree one (Figure 3.6), $t_{1/2}$ was 0.56 hours, for tree two (Figure 3.7), $t_{1/2}$ was 0.59 hours and for tree three (Figure 3.8), $t_{1/2}$ was 0.38 hours. In each case, it was discovered that there was a new steady state, with the level of CO₂ respired by the soil falling below the baseline and finding a new baseline without the root respiration (as roots have been removed from the system).

Table 3.2. Model parameters for fitting a single first-order kinetic equation with asymptote following the grubbing-out event of a *Populus* spp.

Tree	Y_0	a	k	r^2
1	0.362	0.979	1.237	0.73
2	0.383	1.183	1.165	0.87
3	0.338	1.528	1.810	0.88

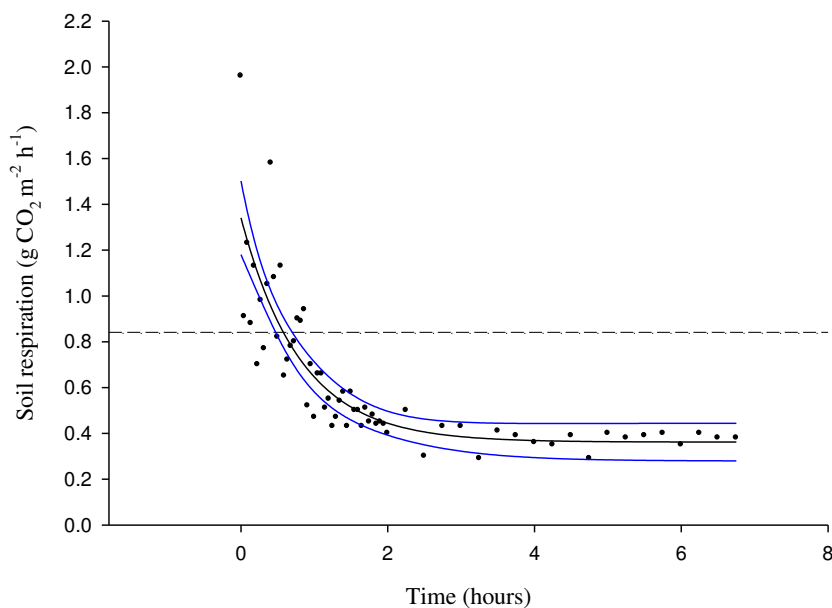


Figure 3.6. The soil respiration following the grubbing-out event of a *Populus* spp. after fitting a single first-order kinetic equation with asymptote. The dotted line represents baseline soil respiration before the grubbing-out event of tree 1. The blue lines represent 95% confidence intervals.

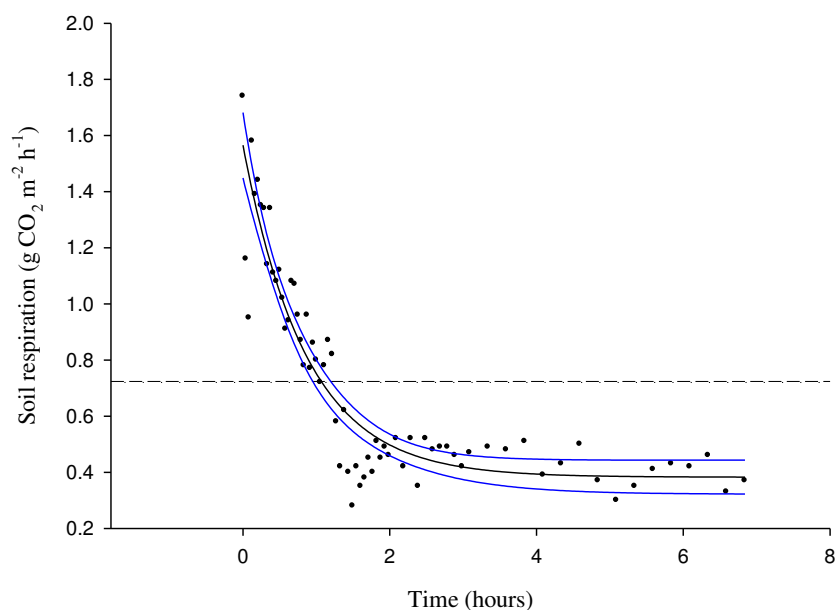


Figure 3.7. The soil respiration following the grubbing-out event of a *Populus* spp. after fitting a single first-order kinetic equation with asymptote. The dotted line represents baseline soil respiration before the grubbing-out event of tree 2. The blue lines represent 95% confidence intervals.

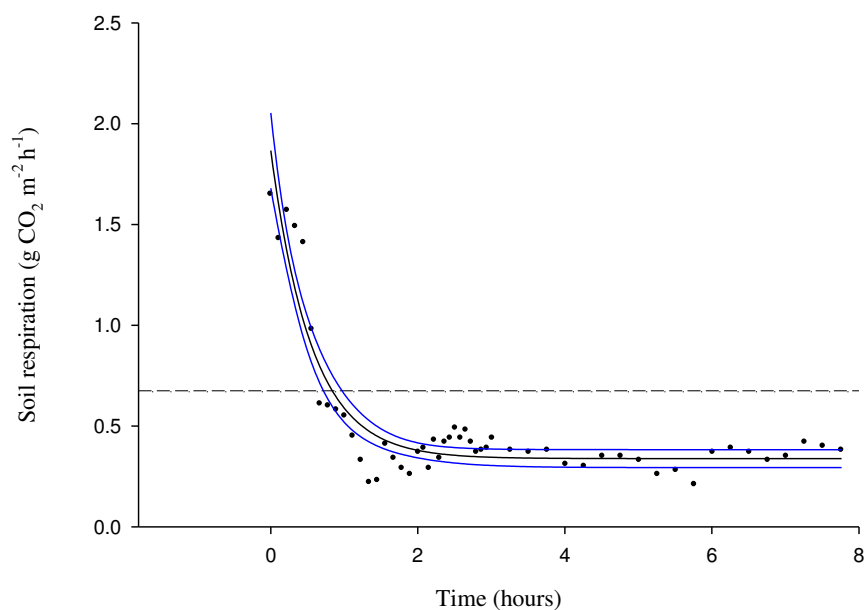


Figure 3.8. The soil respiration following the grubbing-out event of a *Populus* spp. after fitting a single first-order kinetic equation with asymptote. The dotted line represents baseline soil respiration before the grubbing-out event of tree 3. The blue lines represent 95% confidence intervals.

3.4 Discussion

Disturbance to the soil and the subsequent loss of C after its oxidation to CO₂ is known to be one of the three main sources of CO₂ emissions in agriculture (West and Marland, 2002). The sites in this study show that there was a large release of CO₂ from the soil immediately following the grubbing-out process. There was a large peak of CO₂, which was then partitioned to determine exactly where the CO₂ was being lost from and how significant the loss was in terms of the SOC pool and the global C cycle.

The grubbing-out management practice disrupts the steady state of the soil and as the trees are removed from the system, there is a very rapid loss of CO₂, which lasts for approximately 30 minutes. This is a very short response and not a big concern compared with longer term effects of not having trees at all and loss of stable organic matter. A new steady state is then achieved, which is below the baseline. This initial drop of ca.50% is due to the loss of root respiration and supports current data as it has previously been estimated that the contribution of root respiration is between 50% and 60% (Nakane *et al.*, 1996; Hanson *et al.*, 2000; Epron *et al.*, 2001; Epron, 2009; Bond-Lamberty *et al.*, 2004b).

It is expected that stable SOM is now being lost as there are no new C inputs (due to the trees having been removed). However, there will be a small increase in soil CO₂ emissions as any roots and mycorrhizas that have been left behind break down in the soil. Some months after the disturbance event, there will be a progressive decline in soil CO₂ respiration as the microbial C supplies become limiting. This will continue until the ground is prepared for the next crop. Weed growth is controlled on site by herbicide application and although there are grass strips between rows, there is now grass growth on the tree row, thus grass root respiration is not considered when measuring efflux from the grubbing-out event.

It was determined that a maximum of 4.9 g C m⁻² ha⁻¹, with a mean for Site 2 of 1.2 g C m⁻² h⁻¹ and a mean for Site 3 of 2.0 g C m⁻² h⁻¹, in total is lost from the orchard system, which is higher than figures for spring barley under conventional tillage, with emissions peaking between 0.14-0.15 g C m⁻² h⁻¹ (Ball *et al.*, 1999).

It is important to determine whether the C lost from the soil during the grubbing-out period was a true loss from the soil C store or a short-term CO₂ flux due to the rapid release of CO₂ trapped in soil air spaces (Bauer *et al.*, 2006). The CO₂ lost from the studied orchards was partitioned and it was calculated that approximately 95% of CO₂ lost was due to CO₂ being lost from abiotic degassing, where the air is trapped in spaces within the soil structure. Thereby, not being lost from the SOC pool and having very little effect on the global C budget in the short term. Further work should include monitoring the efflux over a much larger period of time in order to determine if these results also apply to the longer term and to determine if there are any other changes over the long term to the global C budget. The actual amount of CO₂ lost from the soil is very small compared to the amount of C lost from standing biomass when the trees are removed. Thus, the grubbing-out management practice has very little effect on the amount of C stored in the SOC (determined as 80 t C ha⁻¹ for the Abergwyngregyn site; Ahmed, 2011), unless the field is left fallow for a long period of time in which case this could become a potential source of CO₂. The SOM is an important component of soil quality and there has been recent concern over SOM and its role in the global C budget through sequestration of atmospheric C in soil (Doran *et al.*, 1998).

In terms of this study, some in-site variation was observed. Site 3 showed no significant difference ($p > 0.05$) between tree treatments, but, at Site 2, two of the trees (trees 7 and 17) released significantly higher amounts of CO₂ ($p < 0.05$). Using the location map and site observation, it was shown that these two trees were at the end of the rows near the area of high disturbance where all the grubbed-out trees were discarded, therefore these two trees had an increased level of disturbance over the other trees tested.

The time of year resulting in varying temperatures is likely to have an effect on the CO₂ emitted from the soil as temperature will affect soil respiration (Bauer *et al.*, 2006). This is supported by the variation observed by our data, where Site 2 displayed lower CO₂ flux than Site 3 as they were measured in Autumn and Spring respectively (Bauer *et al.*, 2006). Site 2 was measured in Autumn, while Site 3 was measured at the end of Winter/beginning of Spring. Ideally, both would be measured at the same time of year under the same conditions; however, measurements had to be taken when the operation was taking place in the orchard.

It is known that the depth of disturbance plays a role in the CO₂ flux observed (Ball *et al.*, 1999). The grubbing-out of relatively large 22 and 30 year old trees creating a disturbance at each tree approximately 1 m and 1.5 m in radius (respectively) and 30 cm and 40 cm (respectively) deep disturbs a volume much larger than the greatest ploughing depth of 30 cm measured by Ball *et al.* (1999), which could explain why the flux from our data is larger.

Conventional tillage and disturbance to the soil does cause a depletion of the SOC pool (Lal, 2004; Smith *et al.*, 2000). Disturbance is known to increase below-ground respiration leading to an increased flux of CO₂ leaving the soil (Lukac and Godbold, 2011). A change in management practice from plough till to less intensive or conservation tillage can result in the soil becoming a C sink by enhancing C sequestration and reducing CO₂ emissions (Ball *et al.*, 1999; Lal, 2004; Schlesinger, 2000; West and Marland, 2002). While this may be a viable option within agriculture, it is not a possible scenario for orchards. It is important to remember that although the CO₂ flux from the grubbed-out orchards was greater than the flux from the tillage of spring barley (Ball *et al.*, 1999), the grubbing-out events occur on a much less frequent basis than conventional tilling.

While disturbances to the soil and the grubbing-out practice release CO₂ efflux from the soil, a full C analysis is required to determine the total amount of C leaving and entering the orchard system. This will be calculated by including the estimation of crop production, energy use, C emissions for primary fuels, fertilizer and pesticide use, lime treatment, irrigation, electricity consumption and farm machinery (West and Marland, 2002). Full C footprints will be calculated for apple orchards (Chapter Six); this will enable direct comparison to many studies, which include fertilizer production and application along with machinery and fuel costs.

In conclusion, the grubbing-out practice, where orchard trees are removed from the orchard by the roots, released CO₂ from the soil on the day of grubbing-out (approximately 0.01 t C ha⁻¹ over the half hour release period) and led to a loss of C in terms of biomass (up to 25 t C ha⁻¹). Approximately 95% of the CO₂ lost was due to abiotic degassing in the soil and not as a loss of C from the SOC pool. Due to this, although there is quite a high loss of C from the standing biomass when the trees are removed during a disturbance event, there is very little

effect on the global C budget. The amount of C lost may even be lower if there is potential to return the biomass in part to the system through the introduction of biochar as a soil amendment. This is investigated in Chapter Four.

CHAPTER FOUR

BIOCHAR-MEDIATED CHANGES IN SOIL QUALITY IN A BRAMLEY APPLE ORCHARD POT TRIAL

Abstract

The production and application of biochar creates the potential to reduce greenhouse gas (GHG) emissions and mitigate climate change through carbon (C) sequestration in soil. In UK orchards where the apple trees are grubbed out (complete removal including roots and stump) at the end of their productive life, there is potential to pyrolyse the waste biomass and incorporate it back into the soil. In this study, an experimental pot trial consisting of Bramley maiden whips ($n = 17$) was set up with four levels of biochar treatment (0, 5, 25 and 50 t ha^{-1}) for two growing seasons. Soil respiration, tree height and dbh were measured at regular intervals, while soil samples were collected and analysed in June 2011 and January 2013. There was found to be no significant differences in tree height growth ($p = 0.88$), in dbh increase ($p = 0.497$), electrical conductivity ($p = 0.450$), bulk density ($p = 0.393$) and moisture ($p = 0.954$) between biochar treatments. There were significant differences found in pH ($p = 0.005$) and organic C ($p = 0.004$), with the 50 t ha^{-1} treatment having a pH with 0.88 units higher and containing twice as much total C than the control. There were significantly different levels of PO_4 , NO_3 and NH_4 between the treatments ($p = 0.011$, 0.001 and 0.005 respectively), but no significant difference in exchangeable Na ($p = 0.886$). There was a significantly higher amount of available K in the highest level of biochar treatment than in the lower treatments ($p = 0.013$), although it was not significantly different to the control. There were no significant differences in available Ca ($p = 0.10$) or in terms of soil respiration ($p = 0.606$). Although there was a strong positive correlation between treatments for soil respiration ($r^2 = 0.92$).

Keywords: *black carbon, carbon sequestration, charcoal, climate change mitigation, fruit orchard*

4.1 Introduction

4.1.1 Climate change, food security and sustainable agricultural systems

At present, there are not only increasing concerns about the changes in the global climate but also global food security. With a rapidly increasing population there is a growing demand for food resulting in an interest in developing more sustainable agricultural systems (Jones *et al.*, 2012). In conjunction with this, it is important to maintain soil quality and develop methods of mitigating climate change. Agricultural soils have a significant carbon dioxide (CO₂) sink capacity, thus soil carbon (C) sequestration is a significant mitigation option to reducing atmospheric greenhouse gas (GHG) concentrations in the face of the immediacy of the climate problem (McHenry 2009; Paustian *et al.*, 1997; Sohi *et al.*, 2009). One potential option of mitigating global climate change is the pyrolysis of woody-wastes to produce biochar, which is then incorporated into the soil. This biomass conversion may be of global significance as a C mitigation tool through soil C sequestration in response to climate change (Brownsort, 2009; McHenry, 2009; Shackley *et al.*, 2009; Sohi *et al.*, 2009).

4.1.2 What is biochar?

Biochar, also referred to as charcoal or biomass-derived black C, has been used in agriculture for thousands of years in the dark earth soils of the Amazon, where it has been seen to improve the fertility of the land, which is in turn related to the stability of C in the soil (Brownsort, 2009; Lal, 2008, Marris, 2006). Since its discovery in 1879, much research has been carried out into assessing its impact on ecosystem C budgets and any significant impacts that biochar may have on soil quality (Lal, 2008; Marris, 2006). Although the exact composition of those “terra preta” soils in the Amazon remains unknown, it certainly has had longevity in the soil, forming a long-term C store (Sohi *et al.*, 2009). Biochar, a by-product of the pyrolysis of biomass, consists of very high C content (approximately 80%) and must be physically inserted into the soil due to it having a fine consistency (Marris, 2006).

4.1.3 Biochar production

Biochar is produced by the pyrolysis of biomass under relatively high temperatures and in anaerobic conditions. The feedstock, temperature and pressure conditions of the pyrolysis process will determine the yield and characteristics of the biochar produced (McHenry,

2009; Sohi *et al.*, 2009). There are three stages in the pyrolysis process: firstly, moisture loss and that of some volatiles; secondly, unreacted residues are converted to gases, volatiles and biochar, and thirdly, the slow chemical rearrangement of the biochar occurs (Demirbas, 2004). Low temperatures and low oxygen (O₂) conditions are favoured for the production of biochar, as there is a higher C recovery from the original biomass and lower production costs, with higher pyrolysis temperatures yielding approximately 5% less C (Lehmann *et al.*, 2006; Lehmann, 2007; McHenry, 2009). The optimum pyrolysis temperature varies with the feedstock used. The pyrolysis of wood is a highly efficient mechanism of C conversion as biochar is around 80% C, while the original wood consists of approximately 45-50% C (Lehmann, 2007).

4.1.4 Feedstock

The starting biomass used to produce biochar should be a material with high C content such as wood (Demirbas, 2004). Biomass for biochar production must be composed of three main groups of natural polymeric materials: cellulose, hemicelluloses and lignin (Brownsort, 2009). The proportions of these in different biomass types will have an influence on the product distributions when the feedstock is pyrolysed (Antal and Gronli, 2003; Brownsort, 2009). The primary products obtained from the pyrolysis of hemicelluloses and cellulose decomposition are condensable liquids and gas, whereas lignin forms these and solid char products upon decomposition (Brownsort, 2009). The type of feedstock used will also determine the nutrient content of biochar (Sohi *et al.*, 2009).

4.1.5 Benefits of biochar

The addition of biochar can potentially increase the levels of soil organic C (SOC), which has an impact on soil fertility and physical properties. Biochar addition may influence aggregate stability, water holding capacity and cation exchange capacity (CEC) within the soil structure (McHenry, 2009). The application of biochar increases the ability of soil to retain nutrients available to plants in cation form (McHenry, 2009). As the biochar consists of a higher percentage C, the application of it increases the C level within the soil, the “terra preta” soils are recorded to have 2.5 times more C than unimproved soils from similar parent material. The crops grown on “terra preta” soils have been found to have a productivity twice as high as that of nearby soils (Marris, 2006). It has been found that the application of charcoal

increases nutrient stocks in the rooting zone of crops, reduces nutrient leaching and thus improve crop production on acid and highly weathered tropical soils (Steiner *et al.*, 2007). It has potential to increase agricultural productivity and provide farmers with a mechanism of participating in C markets by directly applying C into the soil (McHenry, 2009). This addition of biochar is a source of stable C and has high persistence as it is relatively recalcitrant and has a long residence time in soil (Lal, 2008; Steiner *et al.*, 2007).

Any biomass that is waste, such as crop residues can be used in the pyrolysis process (Marris, 2006), thereby acting in a secondary role of waste removal. There is potential for a crop to be harvested at the end of its productive life (such as orchards that are grubbed out), turned into biochar, ploughed back into the field to sequester C and benefit the following crop to be planted (Marris, 2006). A benefit of the charring process is that using the renewable source of C in biomass to produce biochar, releases energy with virtually no sulphur (S) or mercury (Hg) and very little nitrogen (N) and ash waste (Antal and Gronli, 2003). The process of making biochar can result in the production of biofuel as the pyrolysis of farm waste results in volatile organic molecules being given off that can be used as a basis for biodiesel or turned into hydrogen (H) with the aid of steam (Marris, 2006). During biochar production approximately two thirds of the energy that is lost as heat in the conversion process can be captured and utilised in generating electricity or as a heat source (McHenry, 2009). The pyrolysis of biomass and subsequent application of biochar to soil can potentially form a joint strategy to tackle the issues of C sequestration and renewable energy (Lehmann, 2007). With biochar mass being 70-80% less than the original biomass, biochar production offers a potential reduction in transportation costs of waste disposal (McHenry, 2009).

4.1.6 Factors to consider

A key problem with biochar production (the act of burning releases CO₂ into the atmosphere) is that it does not fit into the framework of the Kyoto Protocol. However, the process may be able to contribute to the Clean Development Mechanism, depending on research to determine whether modern char techniques result in sequestering C in the terrestrial ecosystem (Marris, 2006). Currently, biochar as a soil amendment is not included in agricultural policy, largely due to a lack of understanding of biochar behaviour and effects

it may have on soil quality in temperate ecosystems (Jones *et al.*, 2012). There is also concern over what effects (potentially positive or negative) there will be upon crop productivity, health and the surrounding ecology, which is important when considering biochar as a soil amendment in soils which grow food crops, such as apple orchards (McHenry, 2009; Sohi *et al.*, 2009). There is need for long-term field trials to determine the actual effects of the burial of biochar into an agricultural system as conditions will vary from those in a laboratory (Jones *et al.*, 2012).

There are concerns over the addition of biochar into soils without long-term field trials as during short-term studies, a number of issues have been raised. Biochar application may impact the soil chemistry by changing the way that nutrients interact and bind with the soil. The addition of biochar has been found to raise the pH of the soil, resulting in a liming effect (Sohi *et al.*, 2009). Due to the nature of the biochar, a soil with biochar added to it may absorb more heat as the albedo is changed (Sohi *et al.*, 2009) and has been found by Wardle *et al.* (2008) to stimulate the loss of native soil organic matter (SOM). Jones *et al.* (2011) found that biochar binds to herbicides when applied, thus reducing its availability in microbial communities and resulting in the risk of environmental contamination. This is of considerable concern when applying the soil amendment to food chain crops such as apples. There is also concern regarding the introduction of xenobiotics into the soil (e.g. polyaromatic hydrocarbons; Lehmann *et al.*, 2011).

There is further concern regarding the production of biochar as the process can be highly polluting to the wider environment and have a detrimental effect on human health if the process is not controlled and the by-products captured to provide energy for other processes (Jones *et al.*, 2012; Marris, 2006).

In the case of biochar as a soil amendment being implemented, production of pyrolysis feedstock must be sustainable to avoid any areas being exposed to erosion and soil degradation after the biomass has been harvested (Jones *et al.*, 2012).

While biochar addition does have positive implications, it is important to remember that once it has been added, it remains permanently within the soil (Jones *et al.*, 2012; Sohi *et al.*,

2009). There is also the need to carry out field trials in temperate zones as although benefits such as an increase in crop productivity, have been recorded in tropical soils, it remains unknown whether biochar addition will be beneficial in soils that are of a higher quality.

4.1.7 Aims and objectives

It is common practice in most commercial UK orchards to grub out whole orchard fields when productivity falls and it is no longer financially viable to maintain the trees. Grubbing out is the method where the roots and stumps are cleared and the field is left fallow for a year or two before replanting. This practice usually occurs when the trees are between 15 and 35 years old. This practice has two main implications:

- Woody biomass is not left to stand and form a substantial C reservoir.
- There will be an impact on the soil C reserves

A further concern is what then happens to the biomass after the grubbing out period. It has been observed that many farmers burn it at the side of the orchard. There is potential to utilise this biomass as the feedstock for the production of biochar and subsequently apply it back to the orchard soil (although it is recognised that most orchards would not consider purchasing their own pyrolysis chamber). This biochar soil amendment could be an alternative method of producing a stable form of C derived from the above-ground biomass and return it to the SOC instead of releasing it all to CO₂ to the atmosphere. If this is found to be the case, there is potential application of this mechanism to other biological systems, there are not just implications for UK orchards.

The purpose of the study is to investigate the potential for C sequestration and improvement of orchard soils by converting waste apple wood biomass to char at the end of their productivity cycle and applying it back to the system as biochar.

4.1.7.1 Null hypotheses

- There will be no changes in the soil C content with increasing biochar treatment.
- There will be no changes in the tree growth with biochar application.
- There will be no changes in the soil chemistry with biochar application.
- There will be no changes in the soil respiration with biochar application.

4.2 Materials and methods

4.2.1 Biochar production

Grubbed out apple (*Malus*) wood was collected from a local Anglesey orchard (53°11' N, 4°15' W) and from Abergwyngregyn, Gwynedd, North Wales (53°14' N, 4°01' W). It was left to air dry for 2 – 3 months to achieve a 25% or less moisture level. The wood was pyrolysed in a traditional rotund kiln for 10 hours at temperatures up to 450°C. The char was left to cool for three days and then removed from the kiln before being ground by hand into chips approximately 4 mm in size. Samples of char (n = 3) were dried, ground to less than 2 mm and weighed out into 5 mm x 9 mm aluminium cups for analysis in a LECO TruSpec CN analyser to determine C content of the char, which was found to be 81.4±1.4 % (N content was found to be 0.57±0.004 %).

4.2.2 Pot trial

The pot trial was established in 2010 at Abergwyngregyn, North Wales (53°14' N, 4°01' W). The agricultural top soil is classified as a Eutric Cambisol and has a sandy clay loam texture, this was used to mimic soil found in an orchard. The properties of the topsoil are displayed in Table 4.1.

Table 4.1. Chemical properties of the topsoil before the treatment. All values represent means±SEM (n = 3).

Moisture content (%)	pH	Electrical conductivity ($\mu\text{S cm}^{-1}$)	Ammonium (mg N kg^{-1})	Nitrate (mg N kg^{-1})	Available phosphate (mg P kg^{-1})
34.3±1.7	6.2±0.04	42.7±1.6	0.2±0.1	0.5±0.05	0.9±0.2

The *Malus*-derived charcoal was mixed with the soil by hand at rates of either 0 (control, n = 5), 5 (0.08 kg char added per pot; n = 4), 25 (0.4 kg char added per pot; n = 4) and 50 (0.8 kg char added per pot; n = 4) t ha⁻¹ and placed into 35 litre pots along with 1 year old Bramley maiden whips. This apple variety was chosen due to it being widely grown in the UK. The trial pots were laid out in a randomized block design with 0.5 m spacing between each pot and 2.0 m between rows to mimic an orchard design. The prevailing wind was South Westerly. Fertiliser applications to the soil were made in Spring 2011 after consultation with the orchard manager from Site D (Chapter 2). It was advised that Bramley apple trees should have 90 kg ha⁻¹ nitrogen (N), 20 kg ha⁻¹ superphosphate (P₂O₅) and 125 kg ha⁻¹

sulphate of potash (K_2SO_4), with the N application split into 50 kg ha^{-1} before flowering (end of February/beginning of March) and the second application of 40 kg ha^{-1} during the first half of July. A foliar application of 5 kg urea should be applied every 7 – 14 days during the Spring followed by two applications of 25 kg urea after harvest (during autumn). The amount to be applied to the pot trial was determined by the surface area of the pots (surface area = 0.12 m^2) and the % active ingredient of each fertiliser. During the first week of March 2011, 2 g Nitram (34.5% N), 3 g Sulphate of Potash (48% available K_2SO_4) and 1.14 g superphosphate (21% P_2O_5) was applied to the soil of each pot in the trial. During early June 2011, a further 1.39 g of Nitram was applied to the soil of each pot. An application of 0.01 ml urea per tree was applied throughout the Spring as instructed and a further 0.65 ml urea per tree was applied twice during the Autumn. An application of Vitax organic 2 in 1 pest and disease control for fruit and vegetables was given to the trees mid-April, followed by a liquid seaweed feed approximately 6 weeks later (Ian Sturrock Pers.Comm.).

At the beginning of May 2011, one of the replicate Bramley apple trees within the 50 t ha^{-1} biochar treatment suffered from the adverse wet climatic conditions. By the end of May of the same year, the tree was dead at the roots from possible waterlogging. The tree was removed from the experiment and extra drainage holes were drilled into the pots of the remaining trees to prevent the loss of any more replicates.

Throughout the experiment, crop height and stem diameter (10 cm above the graft) were recorded alongside soil temperature in each pot (to a 10 cm depth) and soil respiration every fortnight where possible (due to bad weather conditions, this was sometimes less frequent) with an EGM-4 IRGA respirometer ($n = 4 \text{ tree}^{-1}$; PP Systems Ltd, Hitchin, UK) until January 2012. Leaf respiration measurements were taken at several intervals over summer 2011 but the IRGA proved unreliable and the data have not been included in this study. Soil samples (0 – 20 cm) were taken in June 2011 and again in January 2013. From each pot, three replicate samples were taken and bulked in a plastic bag. Within 1 h of collection, the samples were taken to the laboratory, sieved to $< 5 \text{ mm}$ and chemical analysis was carried out within 24 h. When the samples were not in use, they were stored at 4°C . Stainless steel bulk density rings (100 cm^3) were used to determine the bulk density on each occasion. The

soil was removed, dried at 105°C (24 h) and the weight corrected for stones. The following was used to determine stone-corrected bulk density:

$$\text{Stone corrected bulk density} = \frac{\text{oven dry weight} - \text{weight stone fraction}}{\text{core volume} - \text{stone volume}}$$

An automated weather station recorded soil temperature at 10 cm depth on the experimental site (not in the pots) and rainfall alongside other weather variables every hour throughout the trial.

4.2.3 Soil analysis

Field-moist soil (1:1 w/w soil:distilled water) was used to determine pH and electrical conductivity (EC). The soil moisture content was determined by drying at 105°C (60 h). Total C and N were determined in a TruSpec CN analyser after being weighed out into aluminium cups. The exchangeable cations; sodium (Na), calcium (Ca) and potassium (K), and available phosphorus (P) were extracted with 0.5 M acetic acid (1:5 w/v). The cation extracts were analysed using a Model 410 Flame Photometer (Sherwood Scientific Ltd, UK) and P determined by the molybdate blue method of Murphy and Riley (1962). Determination of soluble C and N in 1:5 (w/v) soil:distilled water extracts using a TCN-V analyser (Shimadzu Corp., Kyoto, Japan) was unable to be carried out due to machine failure. Available NO_3^- and NH_4^+ were determined colorimetrically using 0.5 M K_2SO_4 extracts (1:5 w/v) following the methods of Mulvaney (1996) and Miranda *et al.* (2001). All of these analyses were performed in June 2011 and were then repeated in January 2013 for comparison.

4.2.4 Data analysis

The response of tree height growth (normality $p = 0.912$ and equal variance $p = 0.880$), dbh increase (normality $p = 0.999$, equal variance $p = 0.538$), \log_{10} moisture content (normality $p = 0.064$, equal variance $p = 0.092$) organic C (normality $p = 0.913$, equal variance $p = 0.841$), % N (normality $p = 0.999$, equal variance $p = 0.931$), PO_4^{3-} (normality $p = 0.654$, equal variance $p = 0.076$), NO_3^- (normality $p = 0.667$, equal variance $p = 0.148$) and NH_4^+ (normality $p = 0.962$, equal variance $p = 0.213$) to the biochar treatment, were compared using a one-way ANOVA and post-hoc Bonferroni test (due to unequal sample numbers).

Where the assumptions of normality and equal variance were not met, a non-parametric Kruskal-Wallis comparisons test was used to determine the response of soil respiration, pH, EC, bulk density, % C, exchangeable Ca, K and Na to the biochar treatment. All statistical procedures were carried out using the statistical package SPSS PC version 20 (SPSS Inc., Chicago, USA), with $p = 0.05$ used as the upper limit for statistical significance.

4.3 Results

4.3.1 Tree growth response to biochar addition

During the trial, similar temperature patterns were displayed for the mean soil temperature (to 10 cm depth) each year (Figure 4.1), however, there were differences in the rainfall patterns for each year (Figure 4.2).

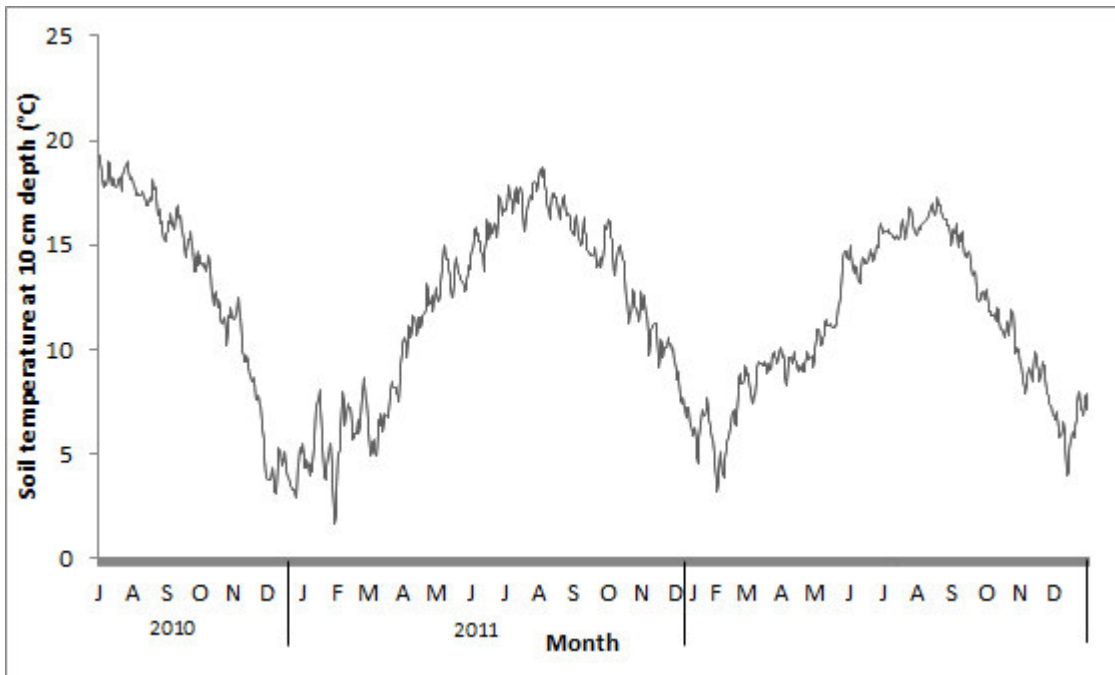


Figure 4.1. Daily mean soil temperature to 10 cm depth at the experimental site for the length of the trial.

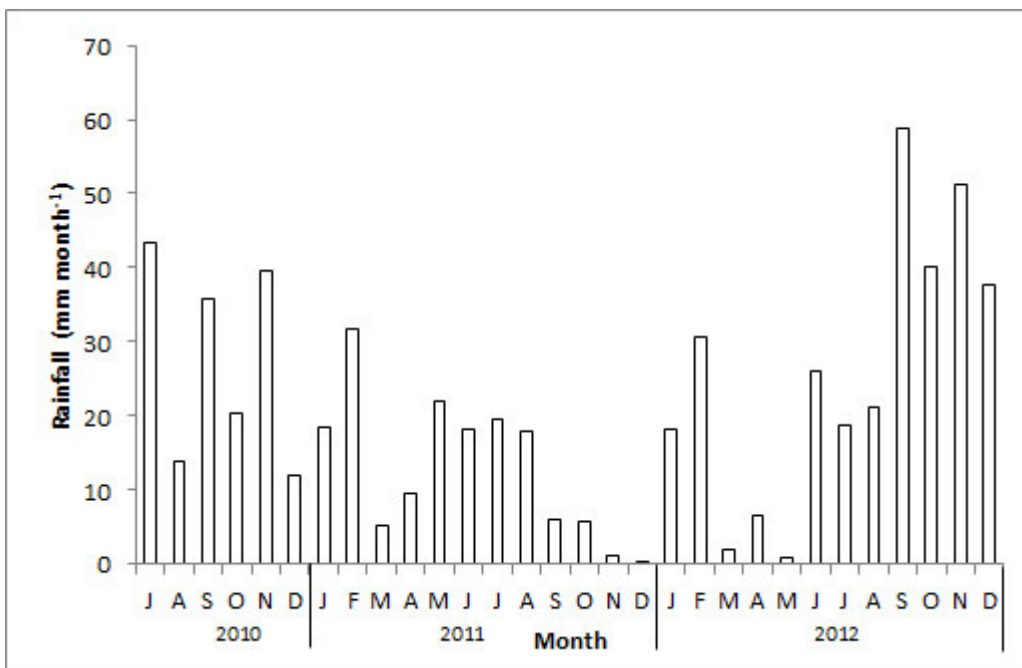


Figure 4.2. Monthly rainfall at the experimental site for the length of the trial.

The response of the *Malus* trees in terms of increase in height is shown in Figure 4.3, with values ranging from 29.6 to 58.5 cm height increase. There is moderate variation within the data for the lowest three treatments, with the coefficient of variation (CV) being 25, 15, 32% for 0, 5, and 25 t C ha⁻¹ biochar treatments respectively. However, there was very low variability for the highest biochar treatment, with a CV of 6.4%. Overall, there is no significant difference ($p = 0.88$) between the treatments, although the trees with 50 t ha⁻¹ biochar appear to have slightly lower increase in height than the other treatments.

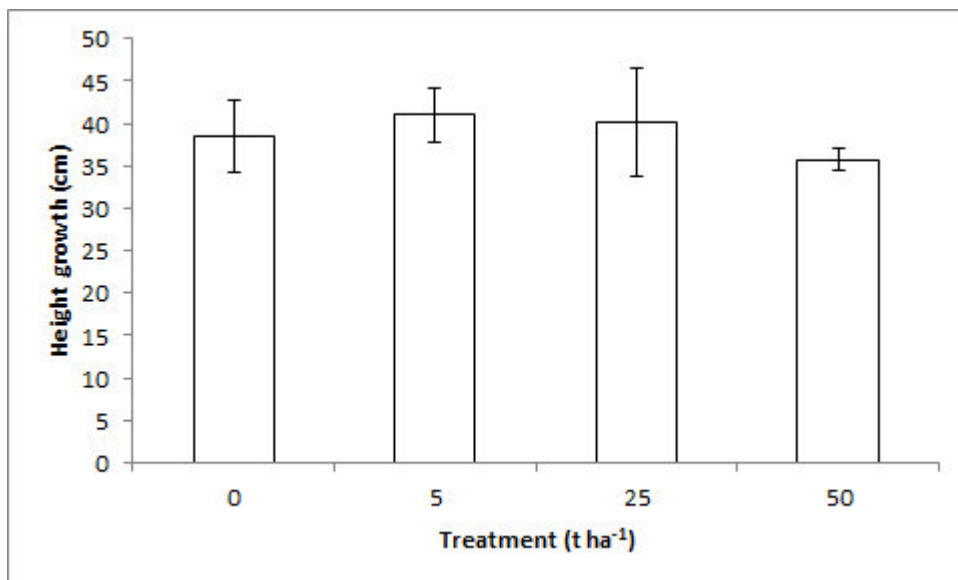


Figure 4.3. Effect of biochar treatment (0, 5, 25 and 50 t ha⁻¹) on the incremental height growth (cm) of the *Malus* trees from the beginning of the trial until the end. All values represent means ($n = 5, 4, 4$ and 3 for each treatment respectively).

The response of the *Malus* trees in terms of stem diameter change is shown in Figure 4.4, ranging from 1.88 to 7.71 mm. There is moderate variation in each of the treatments, with the CV being 40, 39, 20 and 34% for each treatment respectively. With increasing treatment, there is an increasing mean, however, there is no significant difference ($p = 0.497$) between the treatments overall due to the high variability within treatments.

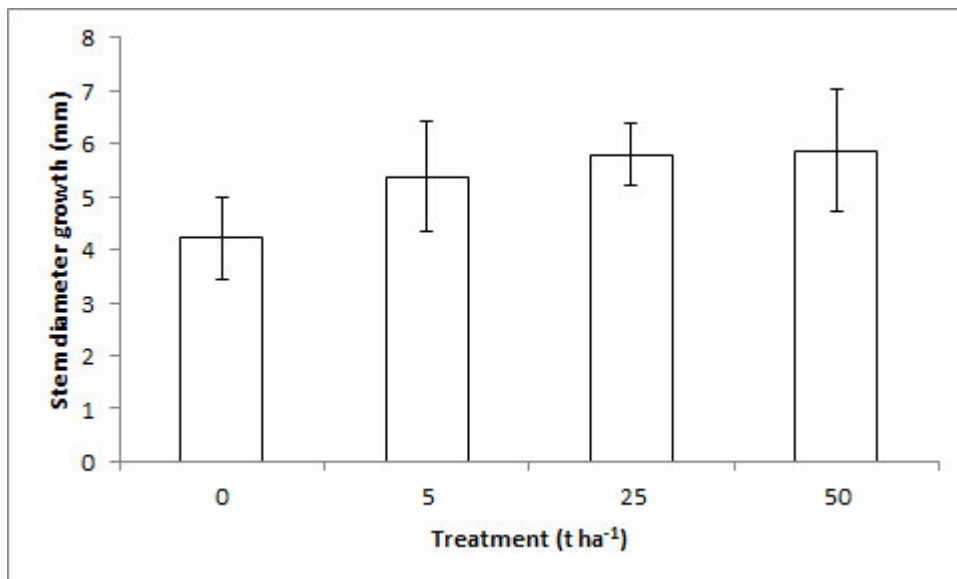


Figure 4.4. Effect of biochar treatment (0, 5, 25 and 50 t ha⁻¹) on the increase in stem diameter (mm) of the *Malus* trees from the beginning of the trial until the end. All values represent means (n = 5, 4, 4 and 3 for each treatment respectively).

4.3.2 Soil responses to biochar addition

The effects of biochar treatment on different soil properties were investigated (Table 4.2). There were no significant differences between treatments in terms of electrical conductivity ($p = 0.450$), bulk density ($p = 0.393$) and moisture ($p = 0.954$). However, there were significant differences in terms of pH ($p = 0.005$) and organic C ($p = 0.004$). The biggest difference in pH was between the control and highest biochar treatment with a difference of 0.88 units in June 2011. The difference between the control and highest treatment was considerably less (0.42 units) in January 2013. Significant differences in organic C were between the highest treatment and the control ($p = 0.013$) and the highest treatment and 5 t ha⁻¹ ($p = 0.021$). There was significantly higher ($p = 0.009$) total C (%) found in the soil with increasing level of biochar treatment. The highest level of biochar treatment held twice as much total C than the control. It was also determined that there were significant differences between the treatments in terms of total N ($p = 0.047$), with the greatest treatment level having significantly ($p = 0.011$) more total N content than the control.

Table 4.2. Chemical properties of the soil at two points in time for each treatment (n = 5 for 0 t ha⁻¹, n = 4 for 5 t ha⁻¹, n = 4 for 25 t ha⁻¹ and n = 3 for 50 t ha⁻¹). All values represent means±SEM, ND indicates not determined.

Sample point	Biochar rate (t ha ⁻¹)	pH	Electrical conductivity (µS cm ⁻¹)	Bulk density (g cm ⁻³)	Moisture content (%)	Organic C (t ha ⁻¹)	Total C (%)	Total N (%)
June 2011	0	5.60±0.10	ND	0.81±0.07	13.63±2.14	12.20±1.14	ND	ND
	5	5.55±0.06	ND	0.93±0.04	12.42±1.42	14.12±0.98	ND	ND
	25	5.69±0.04	ND	0.95±0.04	8.73±2.19	15.74±1.13	ND	ND
	50	6.48±0.01	ND	0.85±0.02	11.49±2.04	15.47±0.75	ND	ND
January 2013	0	6.02±0.02	59.5±4.90	1.01±0.03	38.64±1.87	13.13±0.86	2.49±0.13	0.26±0.01
	5	6.25±0.07	59.2±9.39	0.90±0.04	36.08±1.49	11.29±0.64	2.82±0.17	0.28±0.01
	25	6.36±0.06	71.4±6.46	0.99±0.49	40.60±1.11	14.93±0.55	3.11±0.09	0.28±0.01
	50	6.44±0.12	78.0±5.69	0.96±0.55	42.55±2.89	17.63±2.35	5.02±0.99	0.29±0.01

There was significantly ($p = 0.015$) higher levels of available P between the highest treatment and control ($p = 0.011$) (Figure 4.5). There were also significant differences found for NO_3^- ($p = 0.001$) and NH_4^+ ($p = 0.005$) between treatments. The highest level of treatment had significantly higher levels of available NO_3^- than each of the other treatments respectively ($p = 0.001, 0.002$ and 0.015), while it had significantly higher ($p = 0.003$) levels of available NH_4^+ than the 5 t ha⁻¹ treatment.

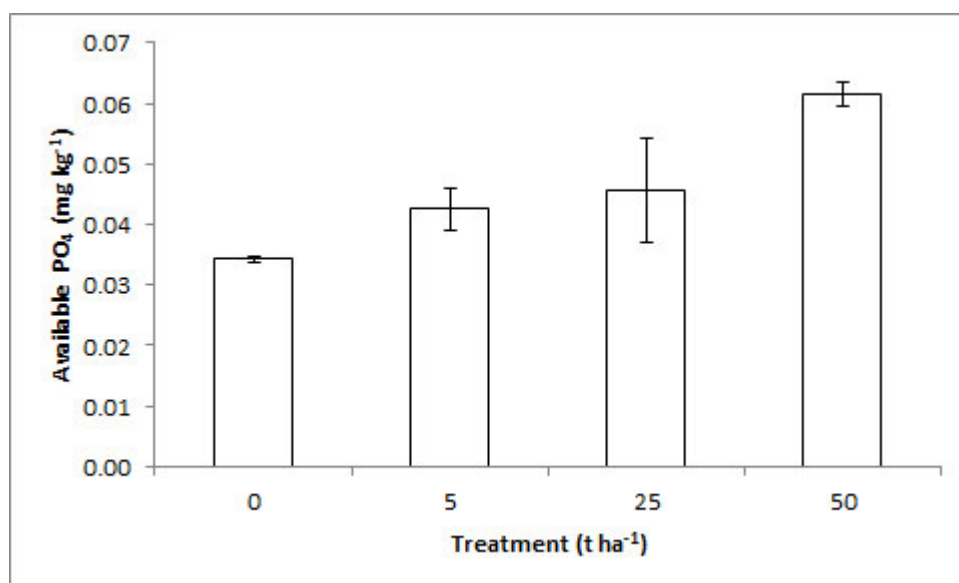


Figure 4.5. The mean values of available P within each treatment (0, 5, 25 and 50 t ha⁻¹). The values represent mean values±SEM (n = 5, 4, 4 and 3 for each treatment respectively).

Although there was a slight increase in the mean value of exchangeable Na observed with increasing treatment (Figure 4.6), there was no significant effect of increasing biochar levels in the soil ($p = 0.886$).

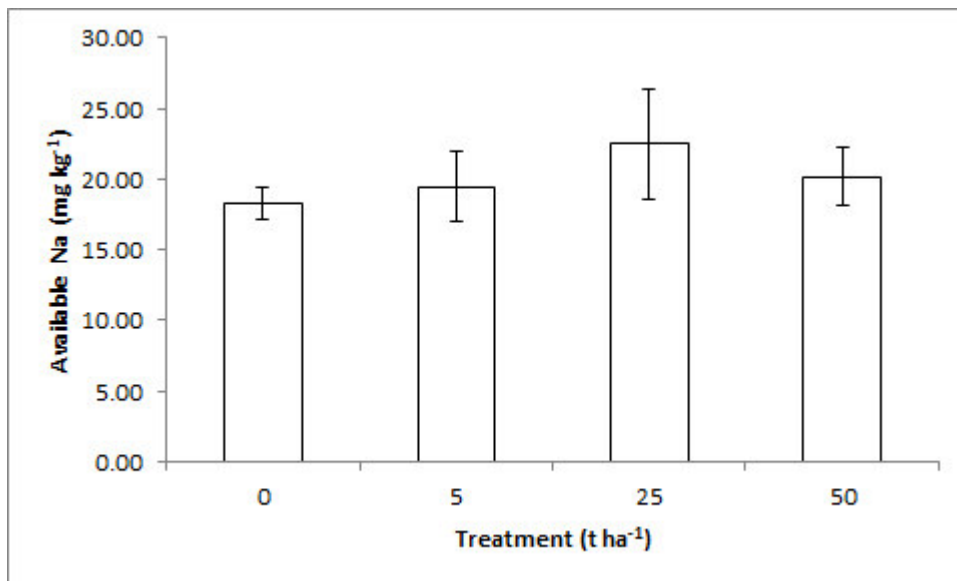


Figure 4.6. The mean values of available Na within each treatment (0, 5, 25 and 50 t ha⁻¹). The values represent mean values ± SEM (n = 5, 4, 4 and 3 for each treatment respectively).

A significant difference ($p = 0.013$) was observed in the levels of exchangeable K found within the biochar treatments (Figure 4.7). There was a significantly lower mean in the 5 and 25 t ha⁻¹ samples (0.06 and 0.10 mg K kg⁻¹ respectively) than found in the control (0.12 mg K kg⁻¹) or the highest treatment (0.16 mg K kg⁻¹), with the 50 t ha⁻¹ biochar treatment displaying a higher mean than any of the other treatments. This trend seems to indicate that there is more available K in the control soil, however, as the soil was all taken from the same place, there is no obvious explanation for the high control value. Although there is a higher observed mean in the two highest biochar treatments, there were no significant differences ($p = 0.10$) in the amount of exchangeable Ca found within the biochar treatments due to the variability in the data (Figure 4.8).

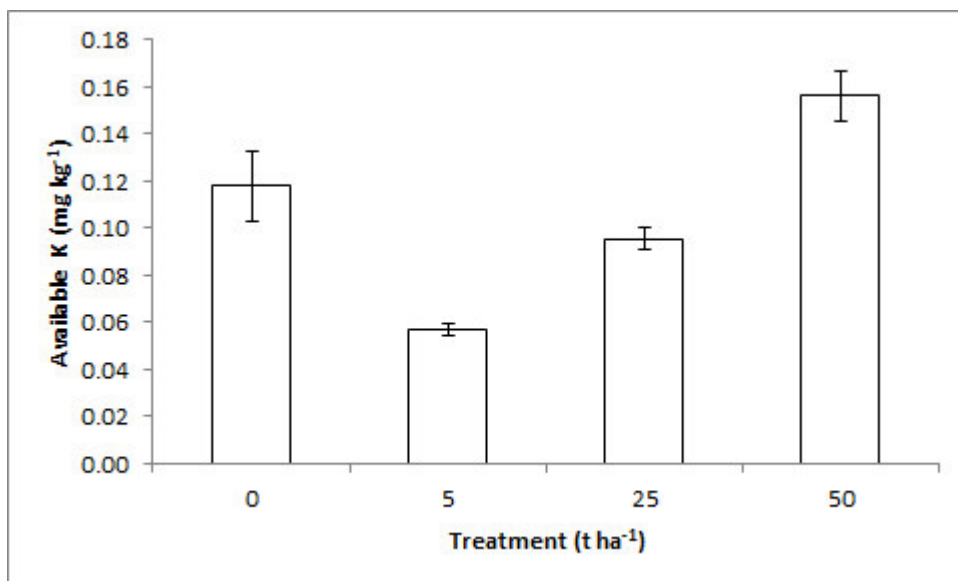


Figure 4.7. The mean values of available K within each treatment (0, 5, 25 and 50 t ha⁻¹). The values represent mean values±SEM (n = 5, 4, 4 and 3 for each treatment respectively).

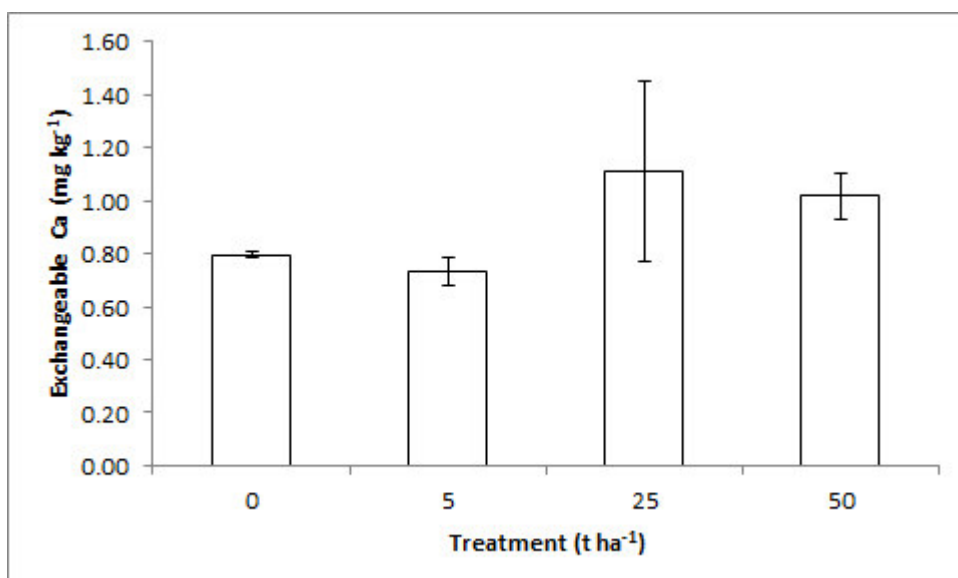


Figure 4.8. The mean values of available Ca within each treatment (0, 5, 25 and 50 t ha⁻¹). The values represent mean values±SEM (n = 5, 4, 4 and 3 for each treatment respectively).

Although there was no significant difference ($p = 0.606$) in amount of CO₂ released from the soil between the biochar treatments, there was a strong positive correlation between soil respiration and biochar application rate ($r^2 = 0.92$; Figure 4.9). These results show that although there was a trend towards increased soil respiration with increasing biochar treatment, the increase was not very big.

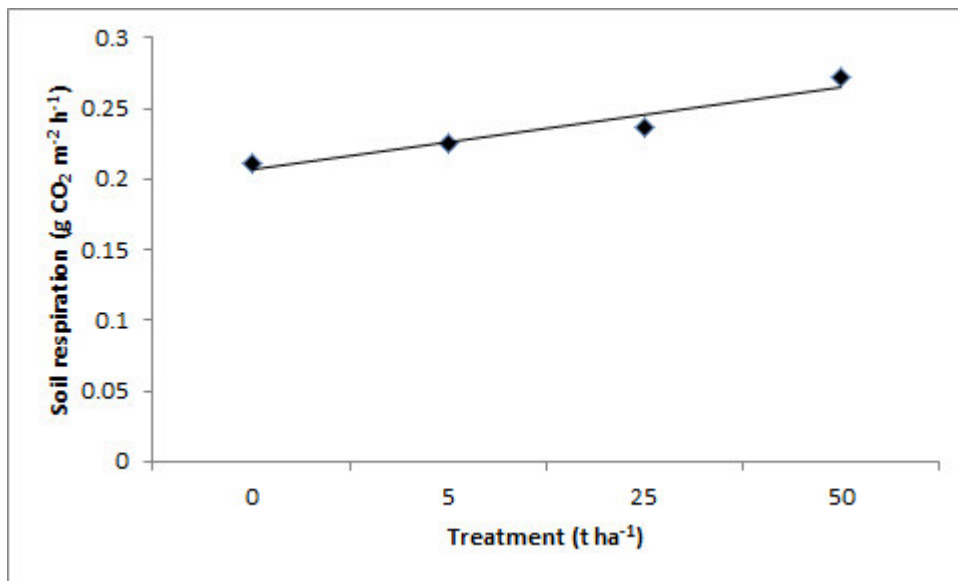


Figure 4.9. Mean values ($n = 496, 400, 400$ and 372 respectively for treatments $0, 5, 25$ and 50 t ha^{-1}) for soil respiration from the start until the end of the trial ($r^2 = 0.92$).

4.4 Discussion

This study determined the effect of planting Bramley apple trees in four levels of biochar treatment (0, 5, 25 and 50 t ha⁻¹), improving understanding of the implications of using biochar as a soil amendment in UK orchards as a response to the grubbing-out management practice.

4.4.1 Tree growth response to biochar addition

The results presented here show that there was no significant increase in apple tree height and no trend with increasing treatment, however, although it was not significant, there was an increasing trend in stem diameter growth with increasing biochar treatment. This may be due to the small sample size, particularly for the highest treatment of biochar and measurements from a larger sample size may give more significant results. The tree growth may not have been affected by biochar due to the rooting depth of the crop. It has been suggested by Jones *et al.* (2012) that crops with a more shallow rooting system (such as common domestic pasture grasses) are affected by biochar addition, whereas deep rooting crops (such as maize or apple trees) are proportionally less affected. As the trial was conducted in pots, there is the possibility that the pot size restricted the tree growth but it is very difficult to tell what the constraints of the pots were and what was occurring in the rooting system. A field trial would enable the tree roots to behave naturally and determine whether or not they are affected by the biochar addition.

4.4.2 Soil responses to biochar addition

It was expected that biochar would increase the pH level of the soil as it has been seen to have a liming effect (Sohi *et al.*, 2009). In this study, the pH at the end of the trial was similar in the control and 5 t ha⁻¹ treatment to the original soil pH before the experiment began (pH = 6.19±0.04) but was significantly higher in the highest level of treatment. This increase in pH caused by biochar additions is what the increased soil fertility is attributed to (Van Zwieten *et al.*, 2010). While in certain soils this may be of benefit, an increased pH level, if it becomes more alkaline than 6.5, would result in growing conditions that may not be favourable to apple trees.

Increased levels of total C within the soil of the highest treatment were as expected due to biochar consisting of approximately 80% C. An advantage of using biochar as a C sequestration method is that the pyrolysis of biomass into biochar results in the labile C being locked into a more stable aromatised form of C (Krull *et al.*, 2009).

Although there were no significant differences between the treatments in terms of moisture content, there was a strong positive correlation between the 5, 25 and 50 t ha⁻¹ treatments ($r^2 = 0.95$) for the final analysis in January 2013. This is in agreement with what was expected as biochar is a porous substance and therefore retains water.

The amount of N is often a limiting factor in plant growth as before it can be taken up it must be mineralised to NH₄⁺ and then nitrified to NO₃⁻ (DeLuca *et al.*, 2009). This study found a significantly higher percentage of total N within the soil of the highest treatment level than found in the control, but this amount was, however, quite small, 0.03%. It was also found that there were significantly higher levels of NO₃⁻ and NH₄⁺ in the highest biochar treatment. This suggests that the biochar is having an effect on the ammonification and nitrification processes on N as suggested by DeLuca *et al.* (2009). This is in contrast with a previous study by Jones *et al.* (2012) who have found that biochar derived from wood has very little effect on N mineralisation or N cycling within the soil. The increase in N found within the high biochar samples may be due to the N present within the biochar itself. As both the ammonification and nitrification processes are biotic, meaning that they are driven by heterotrophic and autotrophic bacteria respectively, it would be interesting to quantify the bacteria and productivity within each of the biochar treatments. There is still a poor understanding regarding the interactions between soil microorganisms, biochar and soil nutrient cycling processes. Similarly, the mechanisms of P availability are not really understood, although pH is known have an effect on availability and biochar conditions have been found to enhance P uptake (DeLuca *et al.*, 2009). This study also found higher levels of available P in the form of extractable PO₄³⁻. This is thought to be due to the addition of P in the biochar itself by an interaction between P being released from the feedstock during the charring process, the sorption properties of biochar and a possible effect of biochar on the environment for microorganisms but these positive effects may be reduced in P-rich environments (DeLuca *et al.*, 2009).

No significant increases in exchangeable Ca or Na were found although the means were slightly higher with the high biochar treatment. The presence of exchangeable K did significantly increase with increasing biochar treatment, again potentially due to the addition of K within the biochar itself, although there were higher levels found in the control than the 5 or 25 t ha⁻¹ treatment. Charring temperature of the feedstock greatly affects the presence and availability of exchangeable cations. Further investigation needs to be carried out to determine optimum temperature for cation availability as increasing the temperature has been shown to increase available cations but if the temperature is too high, the cations are lost by vapourisation (DeLuca *et al.*, 2009). The differences between treatments in terms of soil chemistry could be attributed to a number of factors, such as plant-soil interactions (the plants may have removed more or less of the nutrients) and leaching behaviour (determined by how tightly bound the cations were to the soil or biochar) with varying treatments of biochar addition.

Soil respiration showed no significant increase in release of CO₂, although there is a strongly positive relationship ($r^2 = 0.92$) with increasing treatment. It is not clear whether this increase in CO₂ is a loss of C from within the soil SOM or an abiotic release of CO₂. In order to determine the source of this C loss and its significance, a substrate-induced respiration experiment could be carried out to measure the maximum potential respiration and basal response of the soil.

4.4.3 Limitations of the study

A larger sample size would have yielded more meaningful results, however due to adverse climatic conditions the sample size of each treatment (except for control) was reduced. The trial was carried out in pots arranged in an orchard design and over a short period of time (in orchard terms). Planting the trees directly into the ground and monitoring the response to biochar over an orchard lifetime (between 15 and 35 years) would yield more insight into the reaction of apple trees to the application of biochar to the orchard soil particularly after the initial year or two when apple trees begin to yield much higher amounts of fruit. There would also be no constriction of the root system and nutrient availability from being limited by pot size.

Due to equipment failure, the leaf photosynthesis and respiration was unable to be accurately recorded. This information would give a more complete view of the tree growth response to the treatment level.

A limitation to the study was location and its climatic conditions. Although the soil pH was ideal for growing Bramley apple trees (approximately pH 5.5 - 6.5 is required), the climatic conditions of North Wales are not ideal for Bramley. Welsh varieties would have a better disease resistance and prefer the climate; however, they are not viable for this study as they are not a major apple variety grown commercially in the UK.

4.4.4 Further work

To gain a better understanding of the impacts upon tree growth, future measurements should include leaf respiration and photosynthesis, leaf area index, leaf litter and fruit yield. This will determine if the effects of biochar addition to the soil has an effect on the crop growth and productivity. It would be interesting to apply the treatments to different soil types that are representative of UK commercial orchards to determine if the biochar application has the same effect. Van Zwieten *et al.* (2010) reported that there was a range of responses in terms of biomass production due to both the biochar characteristics and soil type. It would also be useful to study the microorganisms and the effect of biochar treatment on their community and activity. The action of microorganisms is intricately linked with nutrient availability and still little understood.

It would be of interest to determine if the application of biochar to apple orchards would have any effect on the apple fruit itself. As an important food crop, it is imperative to ensure that there are no detrimental impacts on the food chain, which could in turn have a direct impact upon human health. Characterising any impacts on yield or fruit quality would be vital to satisfy the environmental risk assessment and be of interest to orchard owners and policy makers.

The understanding of nutrient-biochar interactions and those of compounds such as herbicides and fertilisers would be increased by the determination of the sorption properties of the biochar. The sorption properties of biochar are of significance in determining the

effect that it will have as a soil amendment (Lehmann *et al.*, 2011). Characterising these properties alongside long-term field trials and environmental risk assessment, will enable a greater understanding of whether using this permanent method of soil amendment will have a positive, negative or neutral effect on real ecosystems. This understanding is key for policy makers to make informed decisions on whether to include biochar in agriculture.

The practicalities of biochar production must also be studied in greater detail, discussions with orchard owners suggested that economic cost would be a great barrier. Some of the sites sampled in Chapter 2 already burn the waste biomass that has been grubbed-out on site and plough it back into the field. However, they would be very reluctant to pay for transportation costs to send the apple wood to be pyrolysed under conditions where the potentially harmful by-products could be collected and re-used for providing energy. It is also unlikely that they would pay set-up costs to establish a proper biomass-burning plant on site. A further area of research would be to quantify the amounts of gases released from the burning of biomass on site at the side of the orchard field and to do an environmental risk assessment on the production of biochar in this manner to determine its viability. Investigations into whether or not repeat applications of biochar to the same soil are viable, would be key to understanding the C sequestration potential. Carbon footprints or Life Cycle Assessment (LCA) should be calculated for the whole biochar process chain to determine the impact this method of sequestration has on the environment, particularly if the orchard owners continue to burn the biomass in open conditions at the side of the orchard.

In conclusion, the application of *Malus*-derived biochar back into the orchard system was found to not yield the same benefits to the growth of the crop as seen in areas with poor soil fertility; however, no negative aspects were apparent in this field trial.

CHAPTER FIVE

EVALUATION OF THE CELL WALL PROPERTIES AND MICROSTRUCTURE OF *MALUS*-DERIVED BIOCHAR

Abstract

With growing concern over the environmental impact of climate change, there is an emphasis on the delivery of mitigation options. One potential solution is the use of biochar; pyrolyzed organic matter, being used as a soil amendment to sequester carbon (C) in the terrestrial biosphere. This study investigated the production and characterization of biochar derived from *Malus* wood. Cubes of wood from a large *Malus* trunk were either left as wood ($n = 27$) or charred ($n = 27$) at 450°C for 15 minutes. A second set of samples ($n = 3$ for each treatment) were charred for varying lengths of time at 450°C from 3, 6, 9, 12 and 15 minutes to determine what effect, if any, length of charring time had on the physical properties of the biochar. Mean cell wall density was the same for both the wood and char samples (1.45 g cm⁻³), with higher variation being displayed in the char sample set. The Brunauer, Emmett and Teller (BET) surface area was significantly higher in the charcoal samples (mean for wood samples = 0.97 m² g⁻¹, mean for the char samples = 5.27 m² g⁻¹). As charring time increased, the mean density increased ($p = 0.016$) and with charring times exceeding 9 minutes, the surface area significantly increased. The charring process has an effect on the material's pore size and the pore size distribution, with the char product resulting in many fewer large pores than the equivalent wood sample. The equilibrium moisture content (EMC) at 95% relative humidity (RH) of the char (7, 15 and 9%) is lower than that of wood samples (21%), with the moisture adsorption isotherms displaying different characteristics. A lack of hysteresis seen in the charred samples suggests that the charring process affects the rigidity of the material, possibly due to removal of cell wall constituents. As the cell wall can no longer expand to absorb water the EMC of the material is reduced.

Key words: *cell wall density, charcoal, dynamic vapour sorption (DVS), hysteresis, isotherm, mitigation option, pore size, properties, structure, surface area, thermal modification, water sorption.*

5.1 Introduction

5.1.1 Background

As concern over climate change increases, there is an emphasis on searching for potential mitigation options. One idea is to lock up more carbon (C) in soil, which already represents a large terrestrial C sink. The incorporation of recalcitrant C in the form of biochar to soil could lock up C that would otherwise be released into the atmosphere, such as when apple orchards are grubbed up, the wood is burnt, which releases carbon dioxide (CO₂) into the atmosphere. If the biomass could be incorporated back into the soil as biochar, some of the C can be sequestered for very long time periods in a quasi-stable state.

5.1.2 Biochar concept

5.1.2.1 What is biochar?

The burning of biomass in a reduced oxygen (O₂) atmosphere results in the formation of ash, bio-gas, bio-oil and charcoal, which can be used for a variety of functions (Brewer *et al.*, 2009). Biochar is defined as the charcoal produced by the thermal decomposition of biomass under anaerobic conditions with the purpose of being used as a soil amendment. It is a porous, fine-grained and C-rich (> 75%) substance obtained under relatively low temperatures (< 700 °C). The International Biochar Initiative (IBI) further defines biochar to have the “purposeful application to soil for both agricultural and environmental gains”. As it has a high amount of C, along with apparent biological and chemical stability, biochar has the potential to act as a C sink, thus playing a role in the mitigation of climate change, through C sequestration (Brewer *et al.*, 2009; González *et al.*, 2009; Manya, 2012; Tsai *et al.*, 2012).

Although the structure of biochar is still uncertain, its porosity is an interesting property as it has implications with regards to water and nutrient retention, pollutant attenuation, and the creation of unique microbial colonization habitats (Kim *et al.*, 2012; Manya, 2012). It is thought that water retention is improved in the presence of biochar and that it subsequently increases the soil surface area (Manya, 2012). Upon addition of the biochar into soil, there is an increase in soil nutrient content due to the addition of nutrients contained within the

biochar itself or due to physiochemical processes allowing better uptake of nutrients from the soil or fertiliser additions (Manya, 2012).

5.1.2.2 Effects of biochar on soil quality

In poor quality or degraded soils, the presence of biochar has been recorded to improve crop yield and the soil productivity and fertility, such as the *terra preta* soils of the Amazon, which due to the presence of man-made charcoal are high in fertility (Brewer *et al.*, 2009; Ippolito *et al.*, 2012; Manya, 2012; Novak *et al.*, 2009; Tsai *et al.*, 2012). It is not yet known whether the same productivity effects will be seen on soil that is already of a higher quality (such as that found within the UK). There is evidence that biochar additions can increase the soil water holding capacity (Manya, 2012, Tryon, 1948). The ability of biochar to improve soil fertility and store C depends entirely on the physical and chemical properties of the char, which will vary between feedstock choice and the pyrolysis conditions (Novak *et al.*, 2009). The reactivity and recalcitrance of biochar in soil is thought to be due to the type of C that forms the biochar (Brewer *et al.*, 2009) and the type of C present in the biochar also varies with the temperature of the pyrolysis process. The temperature conditions have a large effect on all of the properties of biochar and are important to consider when designing the production of biochar. One such example is that of char produced under lower pyrolysis temperatures. The resulting biochar contains aliphatic and cellulose-type structures, which may be good substrates for bacteria and fungi mineralisation, thus playing an integral role in nutrient turnover processes (Novak *et al.*, 2009).

The effects of biochar and its interaction with soil still need further investigation. The studies which have already been undertaken show conflicting results. Spokas *et al.* (2012) looked at 44 published studies on biochar and it was shown that in approximately 50% of cases the plant yield increased and in the remaining 50% either no effect or a negative effect on yield was observed. There is further work to be done in order to find the optimal pyrolysis conditions in order to maximise the benefits of biochar addition to soil (Spokas *et al.*, 2012). There is concern by Joseph *et al.* (2009) that if biochar is applied to soil that has been incompletely pyrolysed, it may have a negative effect on plant yield due to there being bio-available C present but no added nitrogen (N) available for plant uptake. Biochar addition may induce a reduction in soil C mineralization resulting in less available soil N and

sulphur (S) being made available (Ippolito *et al.*, 2012). Low nutrient recovery has been reported in plants where biochar additions have been made to the soil substrate, which in turn leads to a decrease in yield (Ippolito *et al.*, 2012).

Leaching of nutrients from soil is an issue in terms of nutrient availability for plants and it is thought that the addition of biochar to soils may have an influence on this leaching (Ippolito *et al.*, 2012). Ippolito *et al.* (2012) reviewed various studies of biochar use and soil nutrient dynamics and concluded that further research need to be carried out in order to fully comprehend the interactions between biochar characteristics and soil properties in terms of nutrient leaching, retention and immobilisation. Schomberg *et al.* (2012) found that N leaching loss was reduced in some cases but the soil N fraction did not increase with the addition of biochar. These reductions in leaching were thought to be a result of NH₃ volatilization loss from biochars with high ash content. However, a study carried out by Sarkhot *et al.* (2012) found that N leaching levels after biochar application were consistent with those for unamended soil. Jones *et al.* (2012) showed that biochar addition to soil in a temperate agrosystem did not have an effect on dissolved organic C (DOC) and N (DON), NO₃⁻ or NH₄⁺ pools, limited effects on the turnover of plant litter, sugars, organic acids and amino acids and lost its own cations within three years. The study concluded that any changes made to nutrient availability were transient and that short-term positive effects on plant growth found under laboratory conditions were not translated into the field (Jones *et al.*, 2012).

5.1.2.3 Biochar characterisation

The properties of a specific biochar will be determined by the characteristics of the original feedstock and the production process undertaken to produce the biochar (Cagnon *et al.*, 2009; Manya, 2012; Schimmelpfennig and Glaser, 2012; Spokas *et al.*, 2012). The properties influenced by the feedstock include concentrations of elements, density, porosity and the hardness of the resulting char (Spokas *et al.*, 2012). The peak temperature, pressure, vapour residence time and moisture content of the feedstock will all have an effect on the characteristics of the final product as the biomass undergoes structural transformation during the charring event (Kim *et al.*, 2012; Manya, 2012). The peak temperature, highest temperature reached during biochar production, can have an effect on the yield of biochar

produced, the fixed C content, biochar surface area and pore size distribution (Kim *et al.*, 2012; Many, 2012). Kim *et al.* (2012) found that C content rose from 63.9% to 90.5% with increasing temperature (300, 400 and 500 °C) and with the increasing temperature, the C present in the biochar increased in its stability. During the process of pyrolysis the oxygen (O₂) and hydrogen (H₂) is removed allowing the remaining C to form aromatic C bonds (Kim *et al.*, 2012). As the temperature increases, the yield decreases but the product does have an increase in pore properties and fixed C content, particularly when the temperature reached is between 300 and 500 °C (Many, 2012; Tsai *et al.*, 2012). With the increasing pyrolysis temperatures, weak bonds within the feedstock cleave resulting in a lower H₂ and O₂ content, corresponding to the rising C content (Kim *et al.*, 2012).

The macromolecules of the feedstock biomass were found to be weakened by the pyrolysis process resulting in smaller particle sizes with increasing pyrolysis temperature (Kim *et al.*, 2012). Brewer *et al.* (2009) reported BET surface areas of between 7 – 50 m² g⁻¹ and particle density values between 1.5 -1.7 g cm⁻³ for char samples from seven representative char feedstock samples and one commercially available hardwood feedstock. Due to the creation of pores and the cracking of the basal-structural sheets, the surface area of the biochar increases with higher pyrolysis temperatures (Downie *et al.*, 2009). The study by Novak *et al.* (2009) agrees with this as the surface area increased from 0.52 to 1.22 m² g⁻¹ (at pyrolysis temperatures of 400 and 500 °C respectively) for peanut hull-derived biochar, from 1.01 to 222 m² g⁻¹ (at pyrolysis temperatures of 350 and 700 °C respectively) for pecan shell-derived biochar, from 1.10 to 9.00 m² g⁻¹ (at pyrolysis temperatures of 350 and 700 °C respectively) for poultry litter-derived biochar and from 0.40 to 62.2 m² g⁻¹ (at pyrolysis temperatures of 250 and 500 °C respectively) for switchgrass-derived biochar.

Schimmelpfennig and Glaser (2012) recommend that biochar suitable for use as a soil amendment and to increase C sequestration should have specific properties. They suggest the following thresholds: O:C ratio < 0.4, H:C ratio < 0.6, black C > 15% C and a surface area > 100 m² g⁻¹.

5.1.3 Wood properties

It is important to investigate the changes wood undergoes during the charring process to determine the effect of the process on the sample material. The physicochemical properties of wood are mainly determined by three characteristics:

1. The porosity (estimated by measuring the density)
2. The organisation of cell structure and arrangement of cells
3. The moisture content

5.1.3.1 Cell wall composition

The chemical composition of a wood material gives rise to the physical properties present, thus making it an important factor when considering any modification to wood. Hardwoods, such as *Malus* species, comprise vessels, which run longitudinally and allow the transport of water through the trunk. The vessel when cut transversely is exposed and referred to as a pore (Hoadley, 1990). The cells found within wood consist of a thickened cell wall (the primary and secondary layers) and a hollow centre. The main constituents of the cell wall are the macromolecules; cellulose, hemicellulose and lignin molecules, and these are arranged in a precise formation (Bowyer *et al.*, 2002). The cellulose, which is derived from glucose as a result of photosynthesis, makes up between 40 and 50% of the dry mass (Desch and Dinwoodie, 1996). It aggregates and forms long chains, which form a very stable, difficult to break cellulose crystalline lattice (Bowyer *et al.*, 2002; Dinwoodie, 2000). These form the larger structures known as microfibrils, which are long and thin (Bowyer *et al.*, 2002). The cellulose molecules are primarily bonded with covalent bonds, but there are secondary hydrogen bonds arising from the hydroxyl (OH) groups in the cellulose units. These OH groups are highly attractive to water, thus are responsible for the absorption of moisture (Desch and Dinwoodie, 1996; Hill, 2008). Lignin comprises 17 to 33%, thus making it a major constituent of wood. It is a highly complex, non-crystalline molecule which forms a protective layer within the microfibrils, stiffens the wood and makes it difficult for fungi to break the material down (Desch and Dinwoodie, 1996; Dinwoodie, 2000; Findlay, 1975; Higuchi, 1980; Kirk *et al.*, 1980; Sarkanen and Ludwig, 1971). The hemicelluloses comprise approximately 25 to 40% of the dry mass (Desch and Dinwoode, 1996). These components, lignin and hemicelluloses, are thought of as the materials which hold the cell wall together, adding to its structural integrity and stiffness (Dinwoodie, 2000). While lignin acts a matrix,

the hemicelluloses present in cell walls appear to play a role in the adhesion of the cellulose and lignin as they are attached to both, while neither of them have an affinity for the other (Bowyer *et al.*, 2002). Wood also contains a number of highly organic compounds, the extractives, which although are present in relatively low concentrations are of importance both to the material and in economic terms. These extractives are formed in the heartwood and can be easily removed without damaging the structure of the wood (Desch and Dinwoodie, 1996; Dinwoodie, 2000). They are responsible for giving wood its colouration and its durability, due to being comparatively toxic to both fungi and insects (Dinwoodie, 2000).

5.1.3.2 Density

The density of wood is defined as the mass per unit volume and provides a lot of information about the wood under investigation as most mechanical and physical properties are closely correlated to density (Bowyer *et al.*, 2002; Desch and Dinwoodie, 1996; Hoadley, 1990). It has been described as “the single most important property controlling the mechanical performance of a piece of wood” (Desch and Dinwoodie, 1996). The density of a wood will affect the shrinking and swelling behaviour and there is a strong correlation with density and the strength properties, thus making density one of the best criteria for determining the strength of a material along with the physico-chemical composition of the cell wall (Bowyer *et al.*, 2002; Desch and Dinwoodie, 1996; Hoadley, 1981). The strength and stiffness of a wood increase with density, as does the yield of pulp-per-unit volume and the heat transmission of wood. The density of a wood is inversely related to the porosity (or the void volume) of the material, so a low density is expected with a more porous material with density increasing as the porosity decreases (Bowyer *et al.*, 2002). Density may also be used to predict the hardness of a wood (Hoadley, 1981). The density not only refers to the amount of wood substance, but is affected by the presence of both extractives and moisture in the material (Desch and Dinwoodie, 1996). Dry wood is composed of both the solid material comprising the cell walls and cell cavities, which contain air and small amounts of extractives (Desch and Dinwoodie, 1996). Generally, it is assumed that the density of dry wood cell walls (i.e. void free cell walls) is approximately 1.5 g cm^{-3} for all species (Bowyer *et al.*, 2002). Variation in density arises from different materials differing in the cell wall to cell cavity ratio (Desch and Dinwoodie, 1996). These ratios, and ultimately the density of

products, are determined by the proportions of thinner-walled vessels and parenchyma cells and thicker-walled fibres.

5.1.3.3 Surface area

A wide range of porous materials are characterised by the method of gas adsorption, including surface area determination (Shull, 1948; Sing, 2001). The Brunauer-Emmett-Teller (BET) nitrogen method (Brunauer *et al.*, 1938) has become standard procedure in the determination of surface area (Sing, 2001). It is important to measure the pore surface area, as this value gives an indication of how many reactive sites are available within the substance.

5.1.3.4 Moisture Sorption

5.1.3.4.1 Sorption of water

Moisture content (MC) is defined by Engelund *et al.* (2013) as “the ratio of mass of moisture to mass of dry wood substance”. The relative humidity (RH) of an environment is one of the most critical factors affecting the equilibrium moisture content (EMC) of a material (Jalaludin *et al.*, 2010b). In response to the surrounding environment, wood will constantly adsorb and desorb water molecules when it is exposed to water vapour in the hygroscopic range (Engelund *et al.*, 2013; Jalaludin *et al.*, 2010a). Water will be adsorbed when the wood is transferred from low to high humidity environments and desorbed when moved from high to low humidity conditions (Jalaludin *et al.*, 2010b). Under conditions where the RH is stable, the flux of water molecules into the cell wall and the flux outwards into the atmosphere balance. This MC, where the moisture state of the wood is in equilibrium, is known as the EMC (Engelund *et al.*, 2013; Jalaludin *et al.*, 2010a). The hygroscopicity of the material can be reduced with heat treatment, such as thermal modification (Jalaludin *et al.*, 2010a). It is reported that temperatures above 300 °C result in unacceptable degradation to the material and treatment at temperatures of 180 °C or higher will result in chemical changes within the composition of the cell, such as the hemicelluloses and lignin content (Jalaludin *et al.*, 2010a). It is the aim to gain a better understanding of the relationship between biochar and water vapour in order to understand the performance and overall environmental impact of using biochar as a soil amendment.

At a constant temperature, when EMC is plotted against RH of its surrounding environment under a constant temperature and water vapour pressure, the resulting sigmoid curve is referred to as a sorption isotherm (Engelund *et al.*, 2013; Skaar, 1988; Jalaludin *et al.*, 2010b). The uptake of water by a material is referred to as adsorption and the release of water from a material is called desorption (Engelund *et al.*, 2013). The sorption isotherm of cellulosic and lignocellulosic materials (i.e. wood) are characterised by a sigmoidal shaped curve (referred to as a type II isotherm), which typically shows hysteresis between the adsorption and desorption curves (Engelund *et al.*, Hailwood and Horrobin, 1946; Jalaludin *et al.*, 2010a). Whilst it is well known that the sorption isotherm varies between species of wood, it is also recognised that there are differences in the adsorption/desorption behaviour between untreated wood samples and those that have been thermally modified (Jalaludin *et al.*, 2010a; Siau, 1984). The EMC at target RH is greatly influenced by heat treatment, with a marked reduction in EMC being observed with increasing temperature (Jalaludin *et al.*, 2010a).

5.1.3.4.2 Sorption hysteresis

Sorption hysteresis is the term used to describe the amount of water held by cellulosic materials, which is dependent upon the equilibrium relative vapour pressure and the direction that the equilibrium is approached (Jalaludin *et al.*, 2010a). For any given RH, the water adsorbed from the dry condition is always lower than the amount of water retained during desorption (Jalaludin *et al.*, 2010a).

5.1.4 Aim of the study

In the further study of biochar, it is suggested that a two pronged approach be undertaken. On one hand, the production and characterization of biochar needs further investigation, and on the other, studies must be carried out to measure the impact of the additions to both the environment and agronomy. This study is concerned with the former, while the latter is considered in Chapter 4. All impacts, both positive and negative, must be considered if this method of mitigation (the burial of biochar into the soil) is to be adopted. It is important to understand the properties of biochar, and in the context of this study, in particular, the properties of biochar derived from *Malus* wood. Before the method of using biochar as a

soil amendment can be adopted, the effects of adding the char into the system must be understood and the long term consequences considered.

This study focuses on the use of *Malus* wood as the feedstock for the char-making process in response to the burial of the char produced after a grubbing-out event during apple orchard operations. The aim is to characterise the properties, thus determining the suitability of pyrolysing grubbed-out *Malus* trees on site and incorporating the subsequent biochar back into the orchard soil. This process of converting biomass into a long-term store of C, not only addresses the question of mitigating climate change, but also incorporates the idea of waste disposal. It may be possible to incorporate into conventional agricultural production through the utilisation of waste apple trees within the orchard system and converting them into a C sequestration measure.

The aim of the study is to investigate the properties of biochar derived from *Malus* wood by characterising the surface area, density, pore size distribution and volume and to determine the sorption properties of char in order to have an understanding of the absorptive properties of biochar.

The study will also determine if there are any effects on these properties due to charring biomass for different lengths of time.

5.2 Materials and methods

5.2.1 Preparation of char and wood samples

Grubbed out apple (*Malus*) wood was collected from Abergwyngregyn, Gwynedd, North Wales (53°14' N 4°01' W). A 50 cm section of trunk was identified and left to air dry for 2 – 3 months to 25% or less moisture level. It was marked and cut into four transverse layers, then further sub-divided into cubes of roughly equal size (approximately 4 cm x 4 cm). The cubes were marked before cutting and due to the shape of the log (irregular shape with a diagonal edge), some samples were lost (31, 32, 33, 46, 47 and 48). Alternate cubes (n = 27) were charred in a “muffle furnace” (Carbolite 301), while the other cubes (n = 27) were milled to 1 mm using a cross beater mill (Glen Creston). Each cube to be made into char, was sawn in half, then put in a ceramic dish and covered with fine washed silver sand in order to reduce oxidation resulting in the production of charcoal and avoiding the production of ashes during the burning process (Keech *et al.*, 2005). The ceramic dishes were put in the middle of the muffle furnace for 10 minutes to reach 450 °C and for an additional 15 minutes at 450 °C before removal (Keech *et al.*, 2005). Samples were produced at atmospheric pressure.

5.2.2 Preparation of samples for the time series experiment

A section of wood was chosen, that was the same type of wood throughout, from the same *Malus* trunk and divided into cubes for a separate charring analysis to investigate the effect of the charring time on wood properties. Wood samples were converted into charcoal following the Keech *et al.* (2005) method, allowing the sample 10 minutes in the muffle furnace to reach 450 °C with varying lengths of time at 450 °C from 3, 6, 9, 12 and 15 minutes (each set, n = 3). Wood samples were used as a control, only being cut to fit into the testing apparatus (n = 3) and another set (n = 3) of wood samples were milled to less than 1 mm using a cross beater mill (Glen Creston).

Charcoal samples for both experiments were left to cool, brushed to remove the sand and crushed to particles approximately 4 mm (and finer material) in length in order to mimic biochar application as a soil amendment. A vacuum pump was applied to remove all the gas from the samples before further testing.

5.2.3 Determination of cell wall density

Cell wall density was determined for all of the char and wood samples from the *Malus* log and for each of the wood and char samples derived under different lengths of time. In order to calculate the cell wall density, the oven-dry sample was displaced with helium using a helium pycnometer (Micromeritics Accupyc 1330, USA). Helium gas is chosen as due to its inert nature and very small molecule size (approximately 0.2 nm); it can penetrate even the very smallest of pores within the sample without being adsorbed (Tsai *et al.*, 2012). Under helium immersion, the free space within the particles of the sample can be determined (Rossman and Smith, 1943).

5.2.4 Determination of surface area

The surface area of the samples was determined using N adsorbance and the Brunauer, Emmett and Teller (BET) theory based on physisorption (Brunauer *et al.*, 1938; Sing, 2001). Analysis was carried out on dried degassed samples using a surface area analyser (Micromeritics Gemini, USA). N was used as the adsorbate, and to ensure temperature stability, liquid N was used as the sample coolant. In order to calibrate the machine, a Kaolinite calibration standard was used. Micromeritics Stardriver, USA, software was used to determine the surface area using the BET theory, based on volume of N adsorbed at different partial pressures (zeroed for background pressure). The resulting sample values are given as mean values of three different replicates.

5.2.5 Determination of pore size distribution

Pore size distribution is based on the adsorption of a measurable amount of N gas adsorbed by the material calculated from a N isotherm (Sing, 2001), using data from the BET analysis (section 5.2.4), which is then computed in accordance to the Barrett, Joyner and Halenda (BJH) method (Barrett *et al.*, 1951). This method of calculation makes the following assumptions:

- i. That the pores are cylindrical and open-ended
- ii. The radius of the capillary to the relative pressure of desorption is related by the classical Kelvin equation

- iii. Under conditions of the same relative pressure, after the inner capillary desorption, the pore wall has a film thickness that is equal to that of a non-porous surface (Collet *et al.*, 2008).

The following equation was used to determine the pore size distribution:

$$\text{Pore size distribution} = \frac{\text{incremental pore volume}}{\text{average pore diameter}}$$

5.2.6 Determination of dynamic vapour sorption (DVS)

A Dynamic Vapour Sorption Intrinsic (DVS) system (Surface Measurement Systems, London, UK) was used to perform sorption analyses and to determine the EMC of each sample, thus evaluating the sorption properties of both wood and charcoal samples (Jalaludin *et al.*, 2010b). The DVS accurately measures the change in mass of a sample as it sorbs precisely controlled water vapour concentrations within N carrier gas. A small amount of sample (approximately 10 mg) was placed onto a pre-cleaned sample pan which was then carefully hooked onto the hang-down wire connected to the microbalance. The water reservoir in the DVS system was filled with deionised water before the sample chamber was sealed. N was passed over the sample at a flow rate of $200 \text{ cm}^3 \text{ s}^{-1}$ at a temperature of $21^\circ\text{C} \pm 0.2^\circ\text{C}$. The schedule for the DVS was set to start at 0% RH and then increase in 5% steps up to 95% for the adsorption phase and the reverse for the desorption phase. Mass change data were acquired every 20 s. An algorithm was set at each RH to ensure that equilibrium had been reached when the ratio of change in mass in relation to change in time was less than $0.002 \text{ \% min}^{-1}$ for at least 10 minutes of data. The direct monitoring of the actual humidity and temperature was determined by integral humidity and temperature probes located close to the sample. These data were then used in the analysis of the results rather than the set values.

5.2.7 Data analysis

5.2.7.1 Wood and char samples from the *Malus* log

To determine any interactions between the wood and charcoal samples in terms of surface area and density, statistical procedures were carried out using the statistical package SPSS PC version 20 (SPSS Inc., Chicago, USA), with $p = 0.05$ used as the upper limit for statistical

significance. Descriptive statistics were run to indicate the variability of all wood samples and of all char samples. Independent Samples T-Tests were carried out to determine trends in the surface area between wood and char samples and to identify trends in the density of the samples. Both the surface area and density data were assumed to not have equal variance (Levene statistic = 0.00 for both the surface area and density data).

5.2.7.2 Wood and char sample from the time series experiment

To analyse the data from the wood and char samples derived from different lengths of char time, the density data were normal ($p = 0.608$), but did not have equal variance (Levene statistic = 4.217, $p = 0.019$). As one of the assumptions for ANOVA was violated, the non-parametric Kruskal-Wallis test was performed. There is no post-hoc test for samples analysed by the non-parametric Kruskal-Wallis test so the means were compared pairwise in order to determine where differences occur. The surface area data were normal ($p = 0.267$) and had equal variance (Levene statistic = 0.130, $p = 0.982$), therefore a one-way ANOVA and post-hoc Tukey test was used to determine if there were any differences in terms of surface area between the different treatments. All statistical procedures were carried out using the statistical package SPSS PC version 20 (SPSS Inc., Chicago, USA), with $p = 0.05$ used as the upper limit for statistical significance.

5.2.7.3 The evaluation of sorption isotherms

The Hailwood-Horrobin (HH) model was used to analyse the adsorption behaviour of the wood and char samples at each RH using the experimental data (Hailwood and Horrobin, 1946; Jalaludin *et al.*, 2010a; Jalaludin *et al.*, 2010b). According to Harding *et al.* (1998), “the adsorption isotherm is a function of the concentration and distribution of primary adsorption centres, the pore structure, and the vapour pressure”. The HH model considers the state of equilibrium that exists between a vapour, liquid and solid phases and it is as follows:

$$M = M_h + M_s$$

$$= \frac{1800}{W} \left[\frac{K_1 K_2 H}{100 + K_1 K_2 H} \right] + \frac{1800}{W} \left[\frac{K_2 H}{100 - K_2 H} \right]$$

where M is the EMC at a given relative humidity (H), M_h is the moisture content from monolayer sorption, M_s is the moisture content due to polylayer sorption, K_1 is the equilibrium constant of the monolayer water formed from dissolved water and cell walls, K_2 is the equilibrium constant between water vapour and dissolved water, and W is the molecular weight of cell-wall polymer per mole of water sorption sites (Hill, 2008).

5.3 Results

5.3.1 Density and surface area of wood and char derived from the *Malus* log

For the wood samples, there is slightly more variation in surface area (standard deviation statistic = 0.18) than for the density of the samples (standard deviation statistic = 0.004). For char samples, there is slightly more variation in density of the samples than that seen in the wood samples (standard deviation statistic = 0.07) but a lot more variability in the surface area (standard deviation statistic = 8.8).

There is a significant difference between the char and the wood samples in terms of surface area ($p = 0.017$) but there is not a significant difference in terms of density ($p = 0.539$). Figures 5.1 – 5.8 show the surface area and density for each wood and char sample from the *Malus* log. For the wood samples, there is very little variation (coefficient of variance = 0.3% and 18.5% for density and surface area respectively). The mean density is 1.45 g cm^{-3} (ranging from 1.44 to 1.45 g cm^{-3}). The mean surface area for the wood samples is $0.97 \text{ m}^2 \text{ g}^{-1}$ (ranging from 0.68 to 1.36 g cm^{-3}). Although there is very little variation for the density of the char samples (coefficient of variance = 5.0%), there is much higher variation within the char samples for the surface area (coefficient of variance = 166.9%). The mean density is 1.45 g cm^{-3} (ranging from 1.38 to 1.63 g cm^{-3}), while the mean surface area is $5.27 \text{ m}^2 \text{ g}^{-1}$ (ranging from 0.85 to $37.88 \text{ m}^2 \text{ g}^{-1}$).

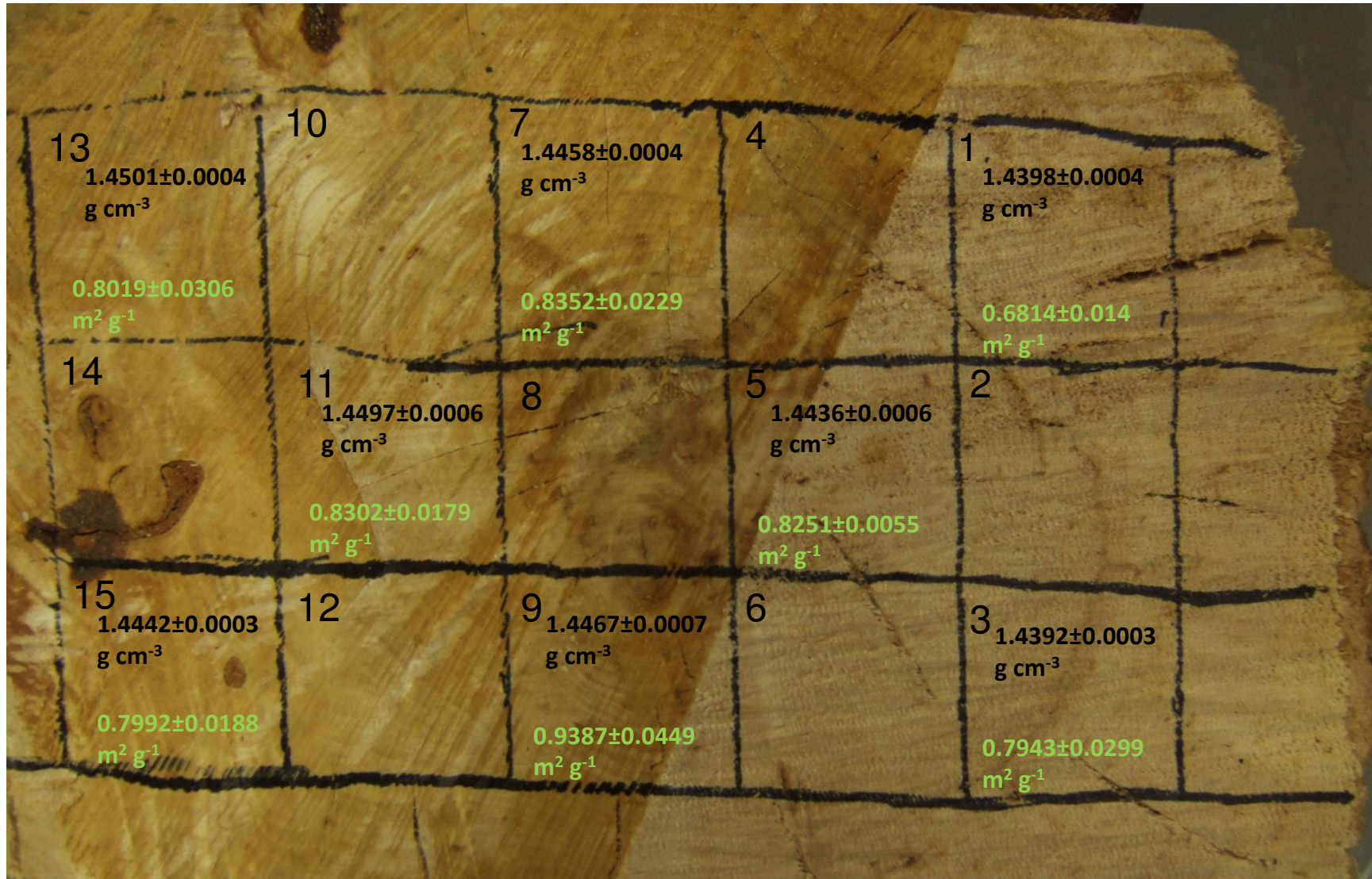


Figure 5.1. Wood samples from the top layer of the *Malus* log (samples 1-15) with the density (black) values and surface area (green) for that sample represented. Scale 1:1.

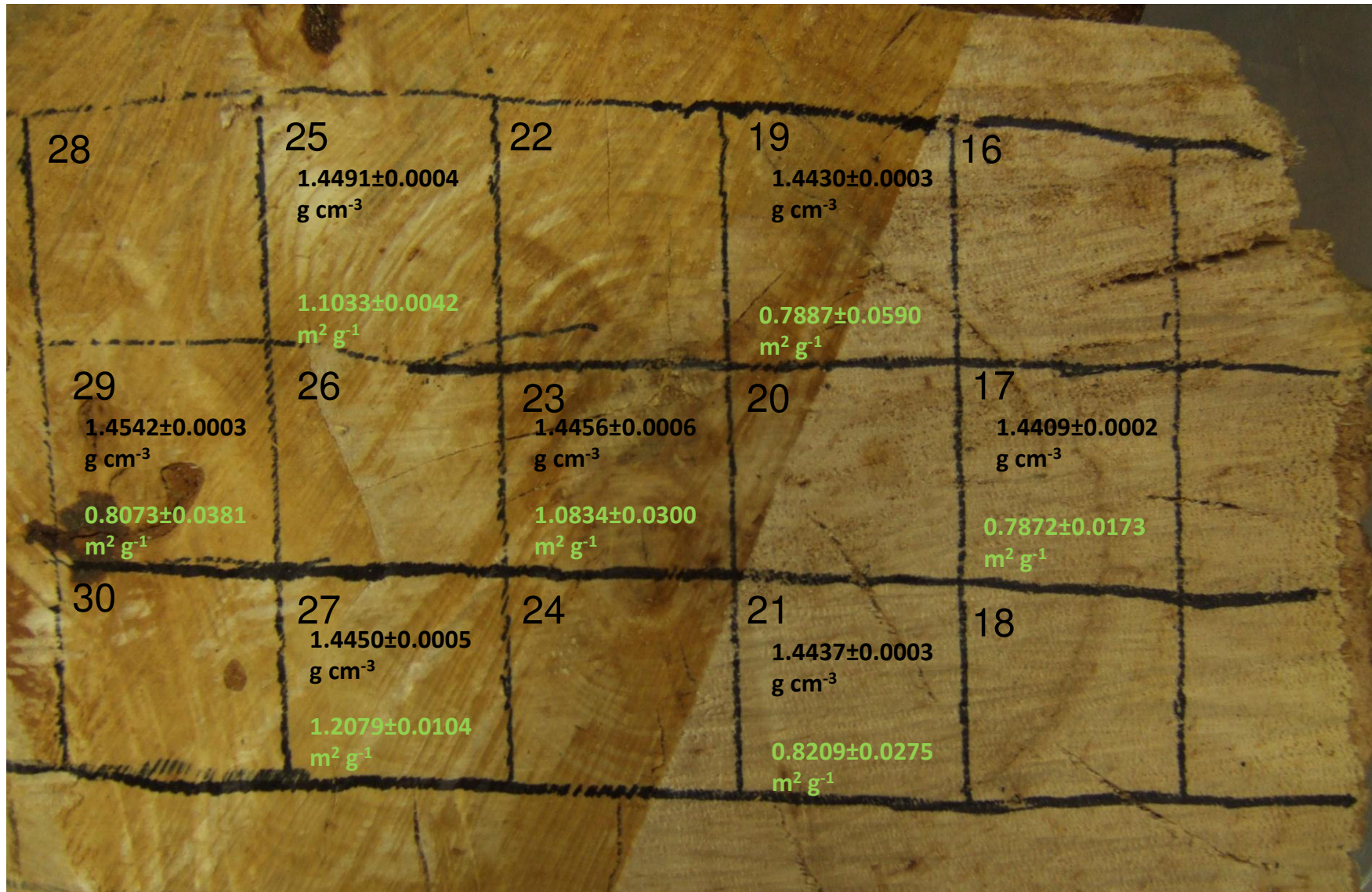


Figure 5.2. Wood samples from the second layer of the *Malus* log (samples 16-30) with the density (black) values and surface area (green) for that sample represented. Scale 1:1.

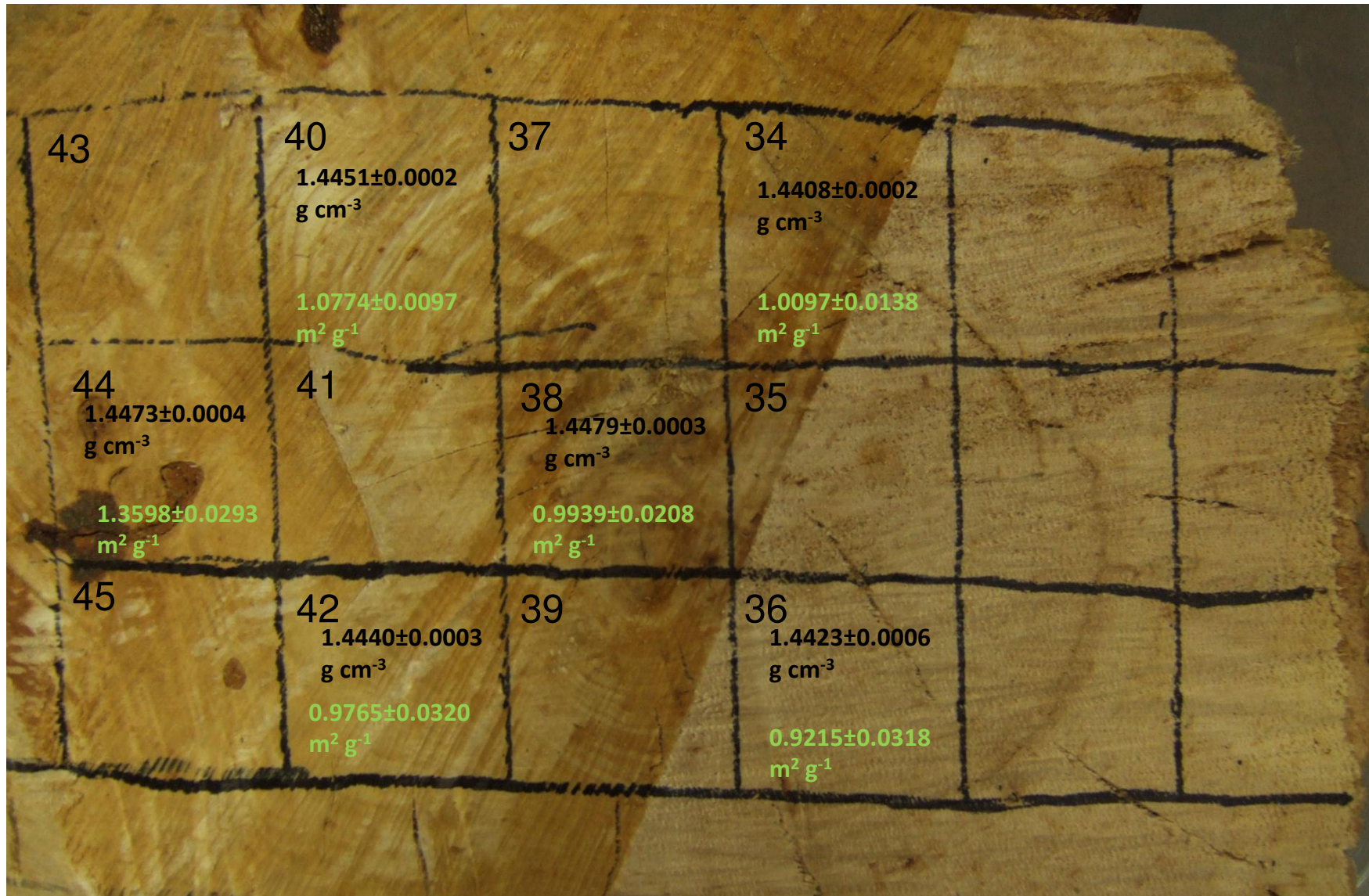


Figure 5.3. Wood samples from the third layer of the *Malus* log (samples 34-45) with the density (black) values and surface area (green) for that sample represented. Scale 1:1.

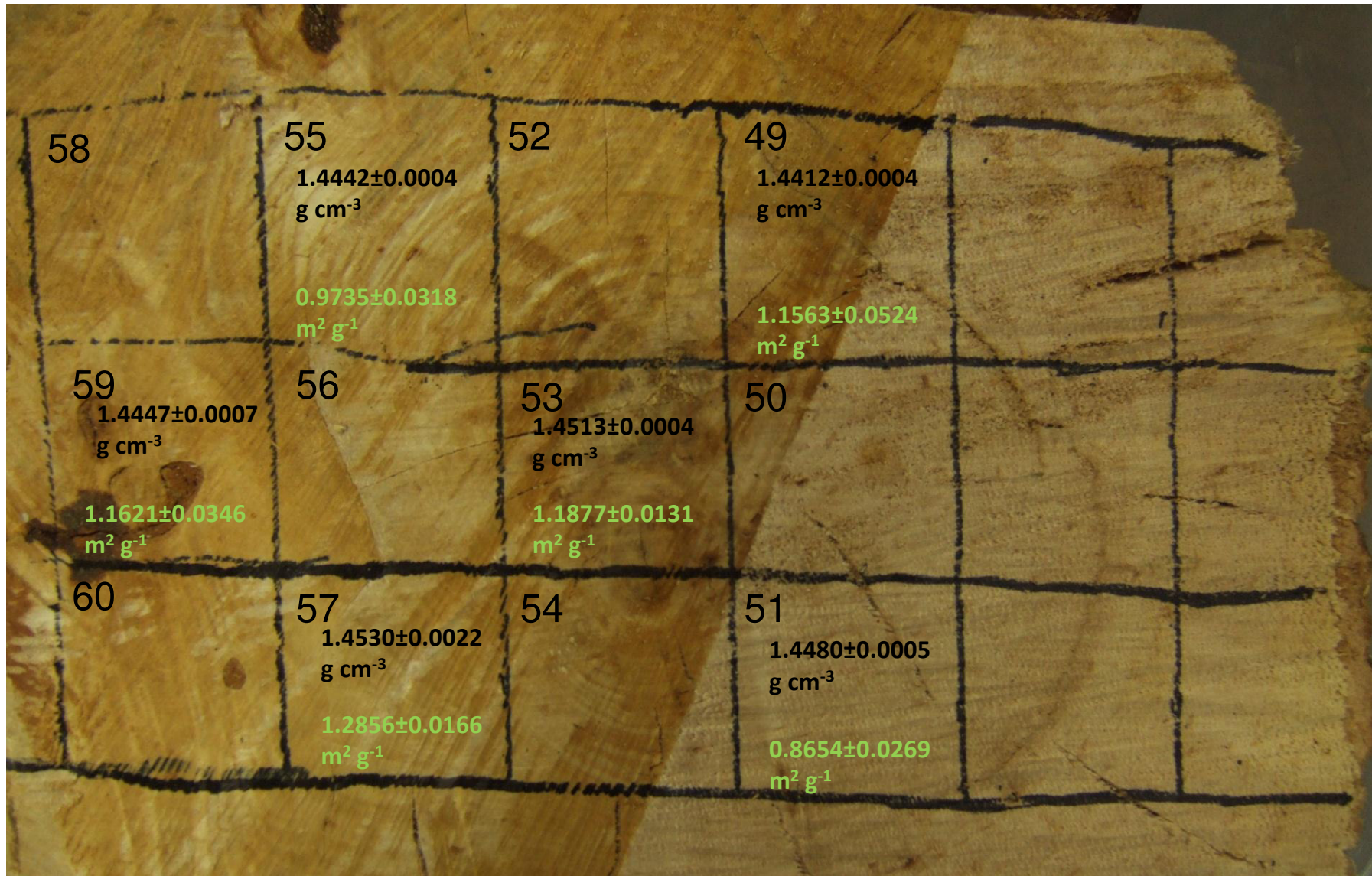


Figure 5.4. Wood samples from the fourth layer of the *Malus* log (samples 49-60) with the density (black) values and surface area (green) for that sample represented. Scale 1:1.

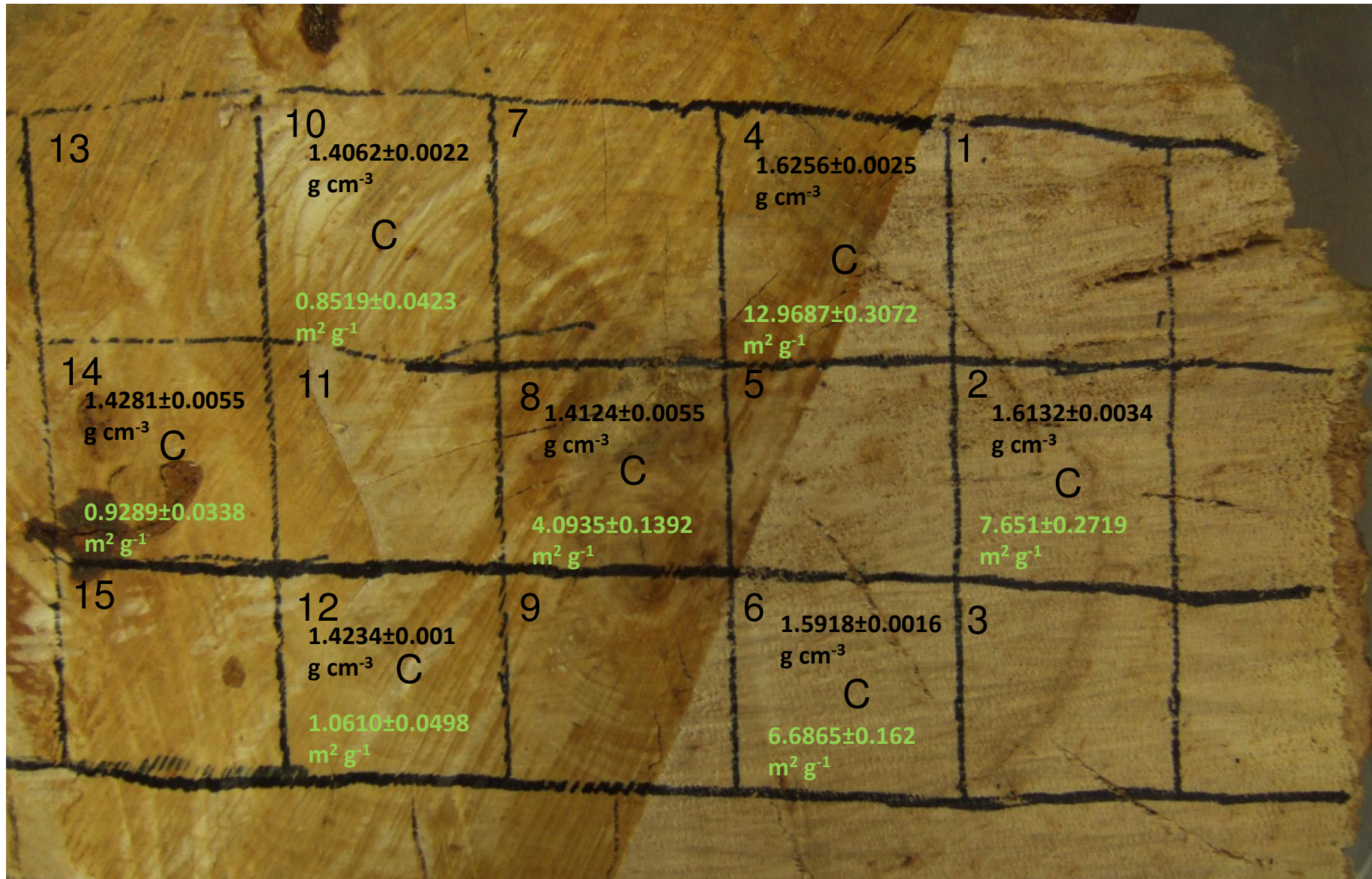


Figure 5.5. Char samples from the top layer of the *Malus* log (samples 1-15) with the density (black) values and surface area (green) for that sample represented. Scale 1:1.



Figure 5.6. Char samples from the second layer of the *Malus* log (samples 16-30) with the density (black) values and surface area (green) for that sample represented. Scale 1:1.

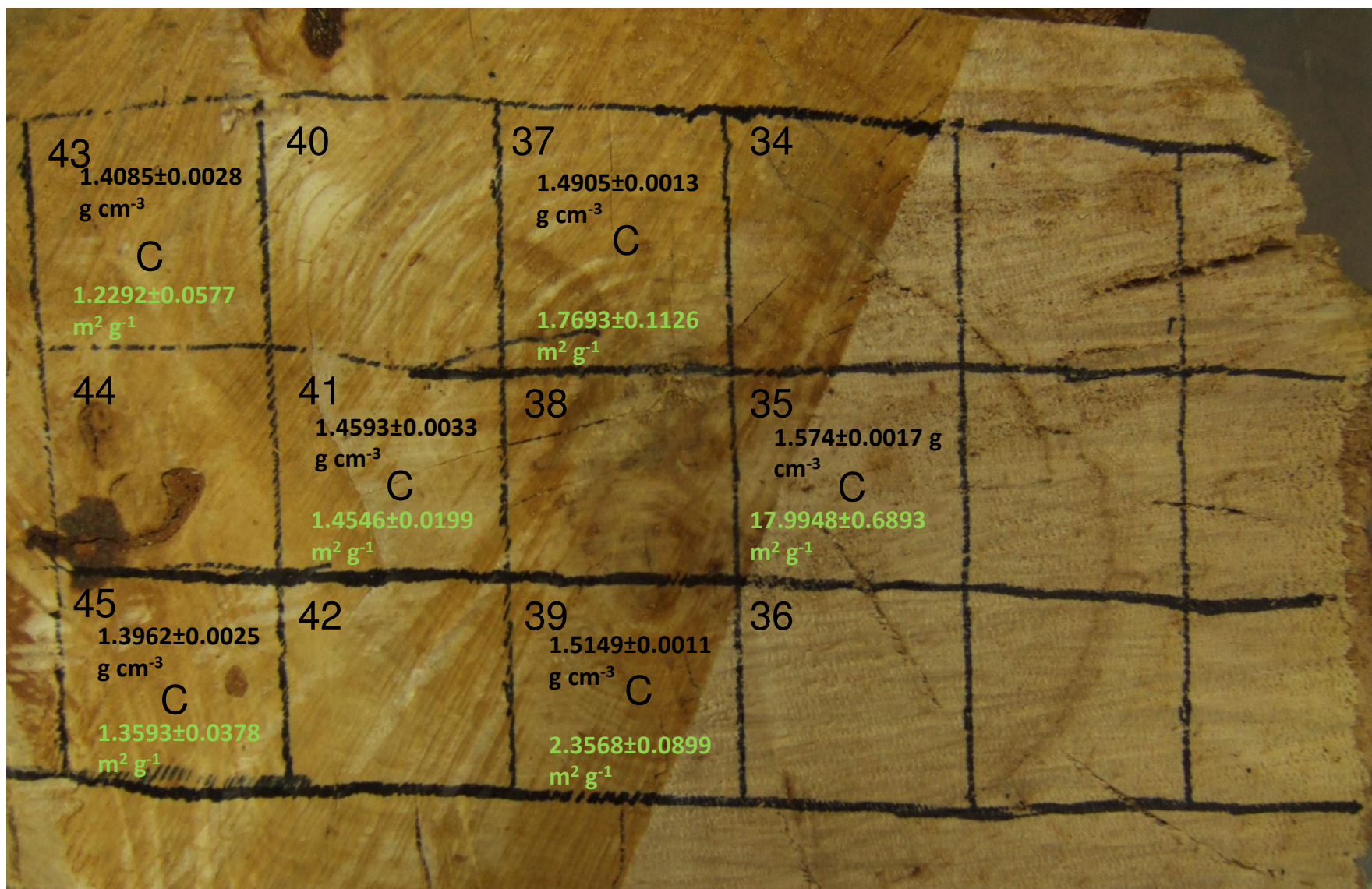


Figure 5.7. Char samples from the third layer of the *Malus* log (samples 34-45) with the density (black) values and surface area (green) for that sample represented. Scale 1:1.

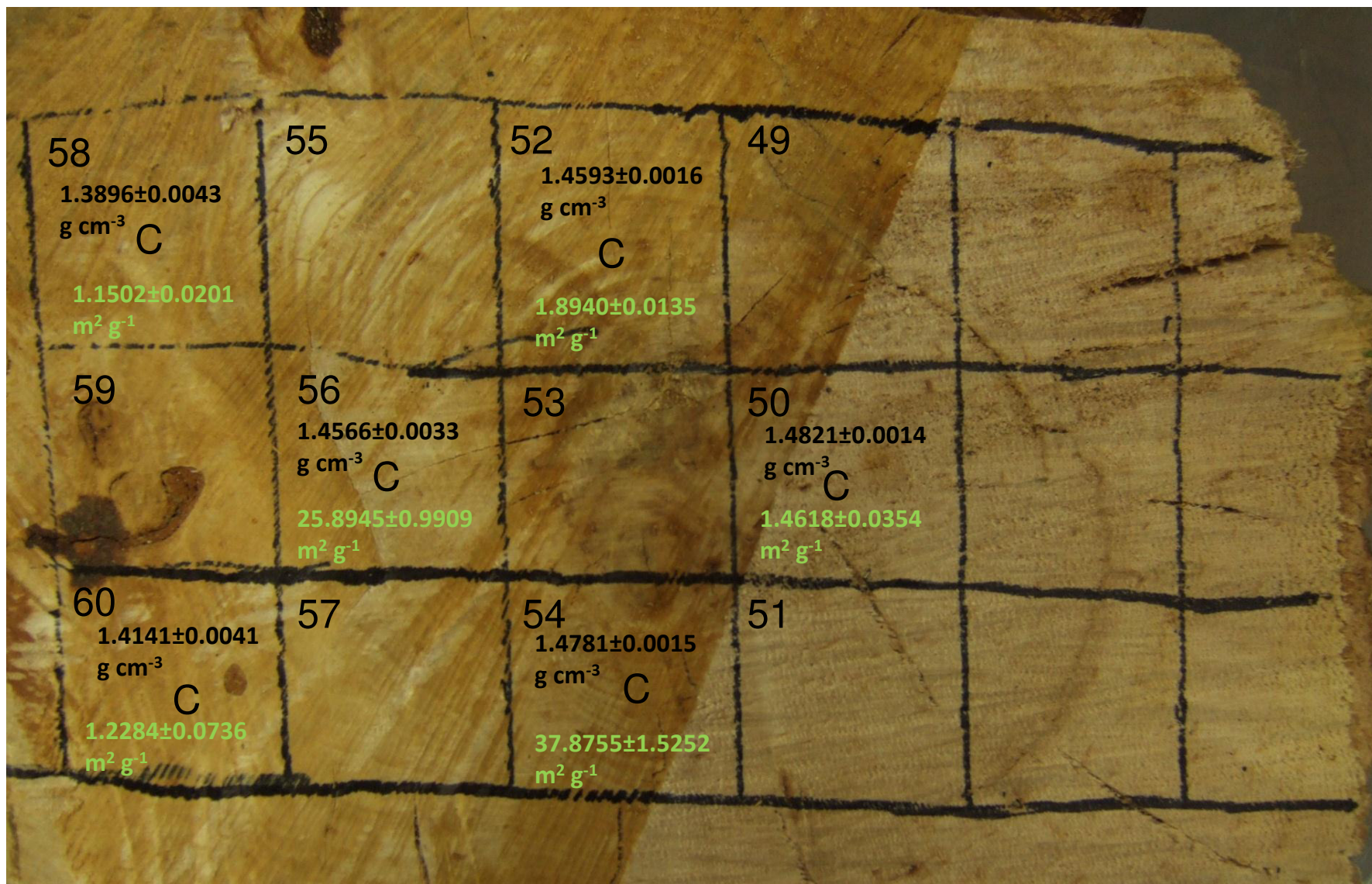


Figure 5.8. Char samples from the fourth layer of the *Malus* log (samples 49-60) with the density (black) values and surface area (green) for that sample represented. Scale 1:1.

5.3.2 Density and surface area of wood and char samples derived from the time series experiment

5.3.2.1 Density

The milled wood sample has a higher mean density (1.64 g cm^{-3}) than the wood left as a block (1.48 g cm^{-3}). Figure 5.9 shows that as the length of charring time increases, the density also increases slightly. There is a statistical significant difference across the lengths of charring time ($p = 0.016$). The mean rank for the density indicates that the milled wood, 9, 12 and 15 minutes charring time have the highest overall ranking with the 12 minutes charring time having the highest rank (15.67).

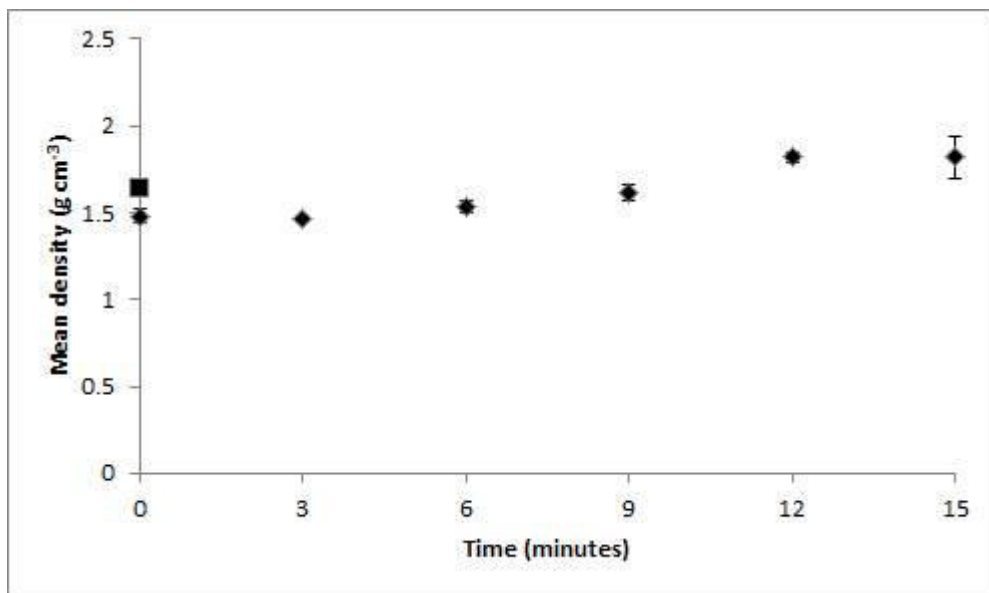


Figure 5.9. Mean density of wood and char samples ($n = 3$) derived from *Malus* log, where time represents the length of charring. At time zero, the square data point represents the milled wood sample, whereas the diamond data point represents the wood sample left whole.

5.3.2.2 Surface area

With regard to surface area, the milled wood sample has a higher mean ($1.30 \text{ m}^2 \text{ g}^{-1}$) than the wood left as a block ($0.54 \text{ m}^2 \text{ g}^{-1}$). Figure 5.10 shows that up until 9 minutes of charring time, the surface area does not change much but there is a steady increase between 9 and 15 minutes. The one-way ANOVA showed that there is a significant difference between one or more of the treatments ($p < 0.001$), with the wood sample having a significantly higher mean surface area than char derived after 3 minutes ($p < 0.001$), 6 minutes ($p < 0.001$) and 9 minutes ($p < 0.001$). There is also a significant difference between the mean surface area of

char derived after 12 minutes than char derived after 3 minutes ($p = 0.002$), 6 minutes ($p < 0.001$) and 9 minutes ($p = 0.003$). Char derived after 15 minutes is also significantly higher in terms of surface area than char derived after 3 minutes ($p < 0.001$), 6 minutes ($p < 0.001$) and 9 minutes ($p = 0.003$). Char derived after 15 minutes was also significantly higher in surface area than char derived after 12 minutes ($p = 0.008$). Figure 5.11 demonstrates the strong linear relationship between charring time and mean surface area after 9 minutes of charring time ($r^2 = 0.9988$).

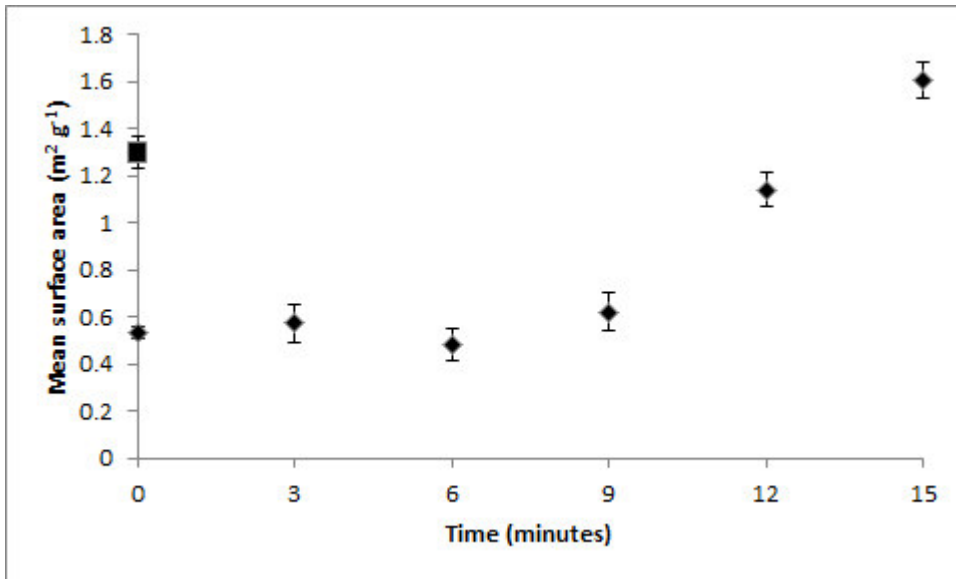


Figure 5.10. Surface area of wood and char samples ($n = 3$) derived from *Malus* log, where time represents the length of charring. At time zero, the square data point represents the milled wood sample, whereas the diamond data point represents the wood sample left whole.

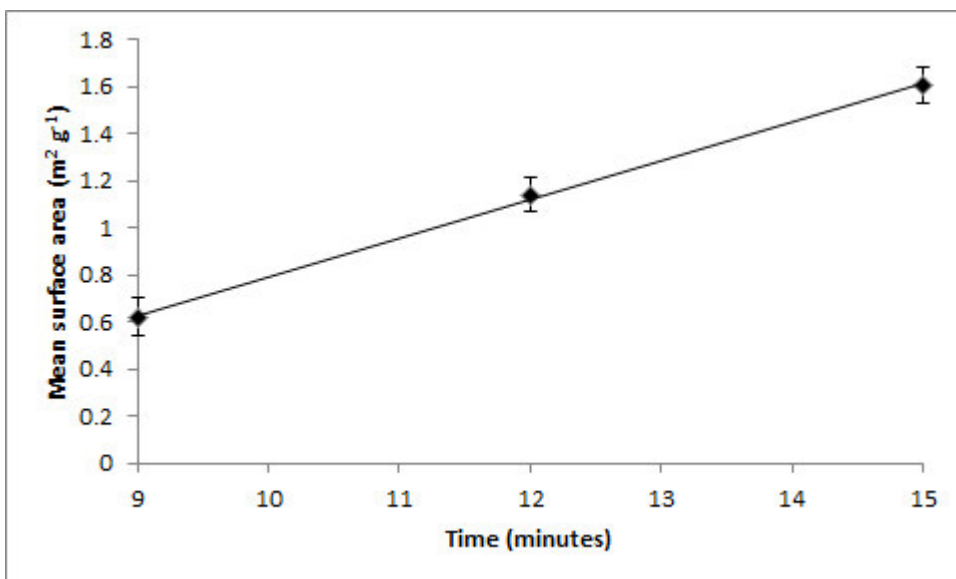


Figure 5.11. Linear relationship between charring time and the mean surface area ($r^2 = 0.9988$) of the three highest charring times ($n = 3$).

5.3.3 Pore size distribution and Incremental pore volume

Figure 5.12 shows a typical pore size distribution of the wood samples tested. The one chosen to represent in graphical form was wood sample 1, the matched pair of char sample 2, which represented a mean surface area. The wood samples have a high number of large pores, while the char samples have many fewer numbers of large pores. This is also shown by Figure 5.13 which shows that the wood sample has a higher number of large pores than the char sample which has much fewer large pores present.

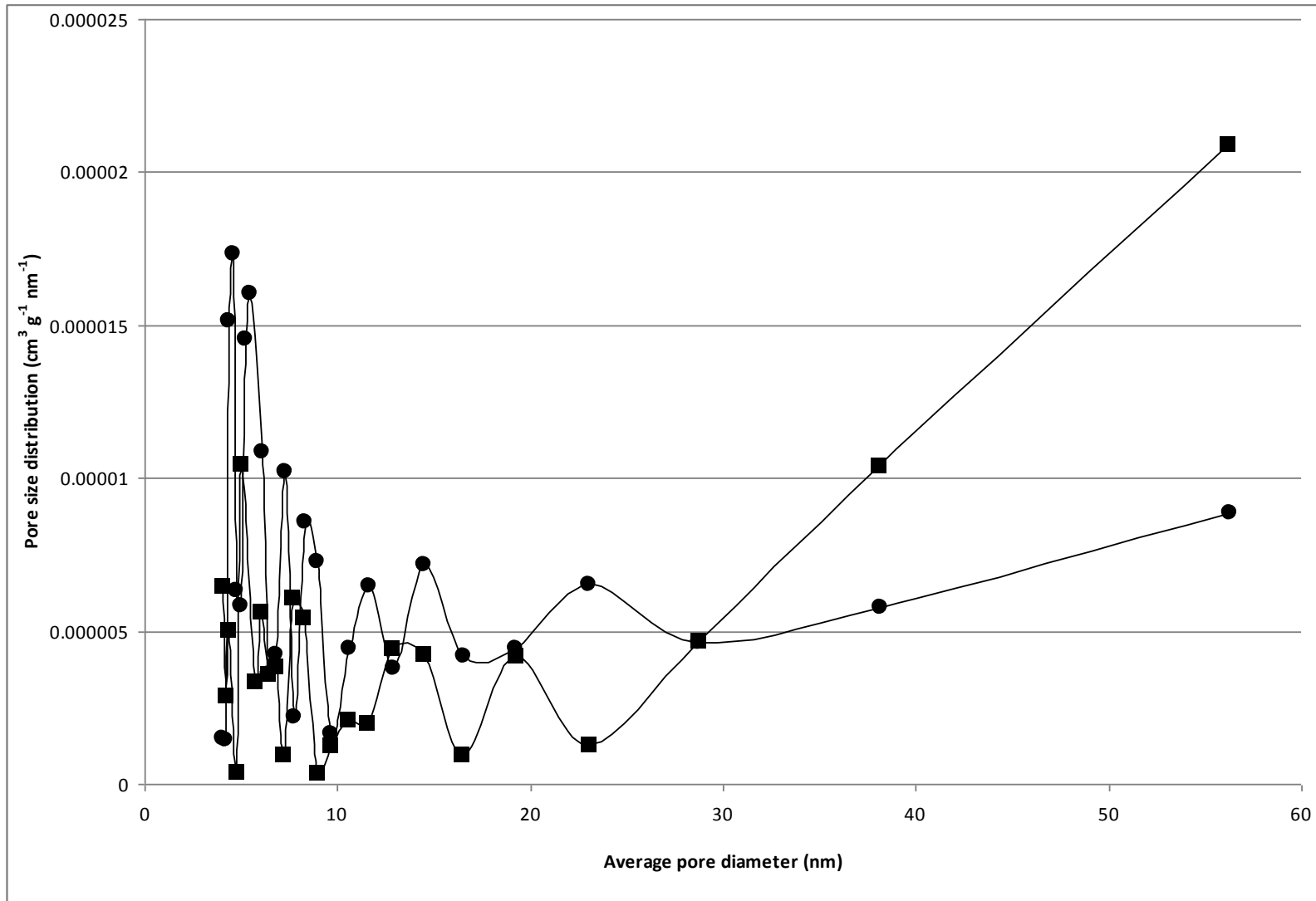


Figure 5.12. Pore size distribution wood (square markers, $n = 1$) and char (circular markers, $n = 1$) for one representative sample of approximately mean surface area values derived from the *Malus* log.

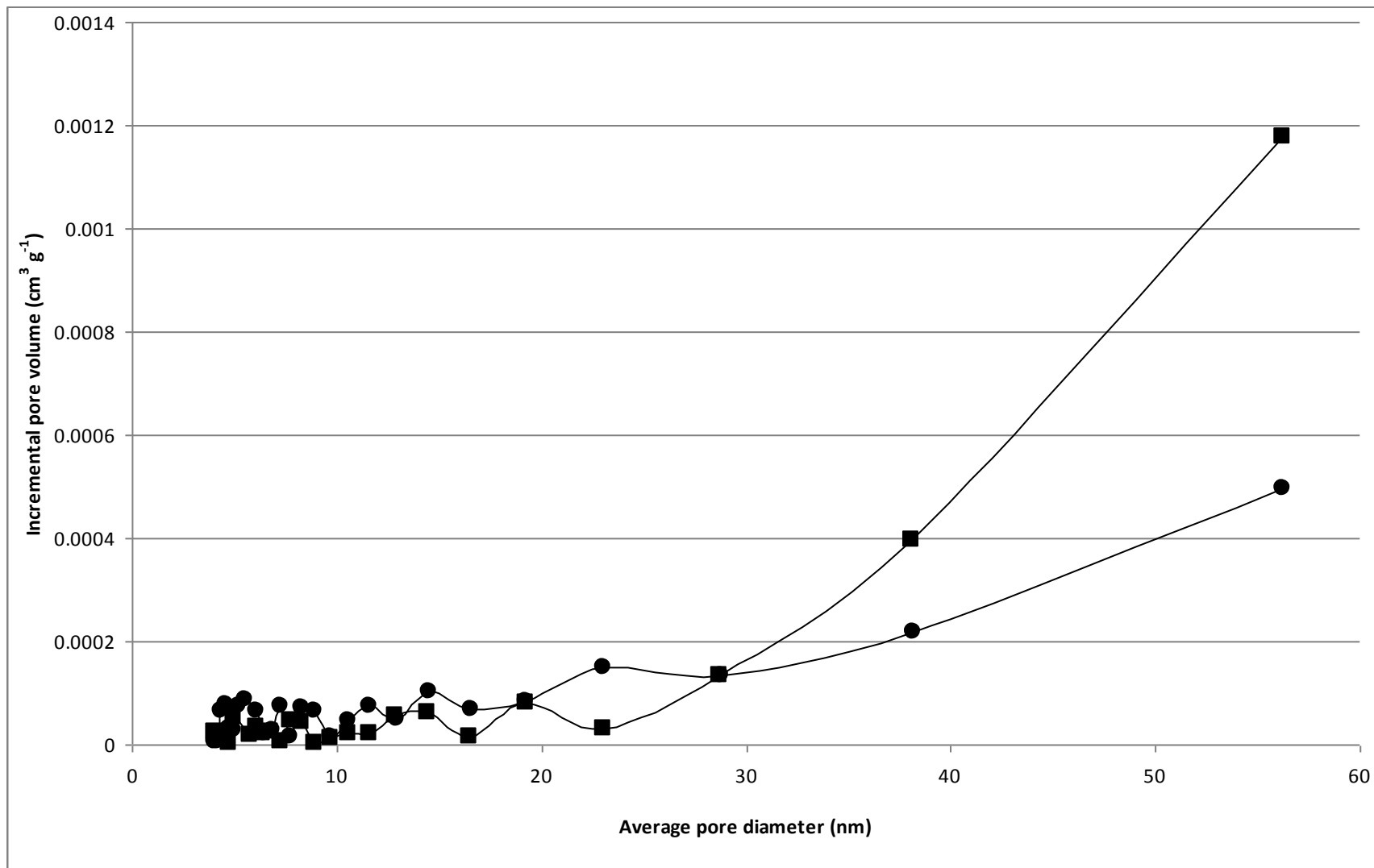


Figure 5.13. Incremental pore volume wood (square markers, $n = 1$) and char (circular markers, $n = 1$) for one representative sample of approximately mean surface area (sample samples as pore size distribution derived from the *Malus* log).

5.3.4 Comparison of sorption isotherms of wood and charcoal

The equilibrium moisture content (EMC) at 95% relative humidity (RH) for the wood sample is approximately 21%, while the EMC changes distinctly with the char samples in Figure 5.14 (a-d). The EMC for the char with low surface area ($0.85 \text{ m}^2 \text{ g}^{-1}$) is approximately 7%, for the char with medium surface area ($12.97 \text{ m}^2 \text{ g}^{-1}$) is approximately 15% and for the char with high surface area ($37.88 \text{ m}^2 \text{ g}^{-1}$) approximately 9%. The moisture adsorption isotherm shows the characteristic sigmoid type II curve with sorption hysteresis for the wood sample (a) but the moisture adsorption isotherms for the char samples display different characteristics. There is very little hysteresis for the char samples, particularly char 5, which has the highest surface area measurement (Figure 5.15).

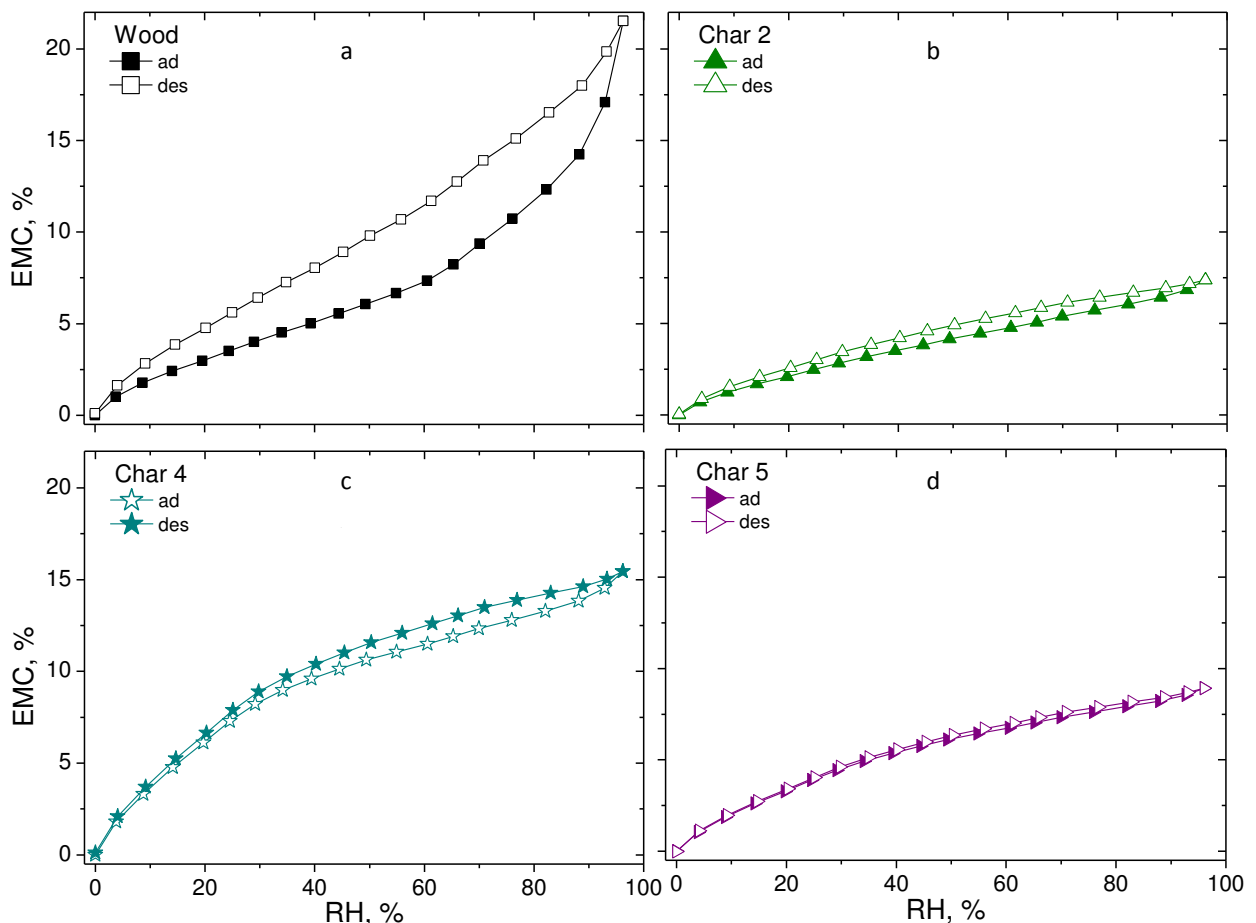


Figure 5.14. Experimental isotherms for *Malus* wood (surface area = $0.87 \text{ m}^2 \text{ g}^{-1}$), char with low surface area (char 2, surface area = $0.85 \text{ m}^2 \text{ g}^{-1}$), medium surface area (char 4, surface area = $12.97 \text{ m}^2 \text{ g}^{-1}$) and high surface area (char 5, surface area = $37.88 \text{ m}^2 \text{ g}^{-1}$). The char was derived from *Malus* spp. under atmospheric pressure conditions and a temperature of $450 \text{ }^\circ\text{C}$ for 15 minutes.

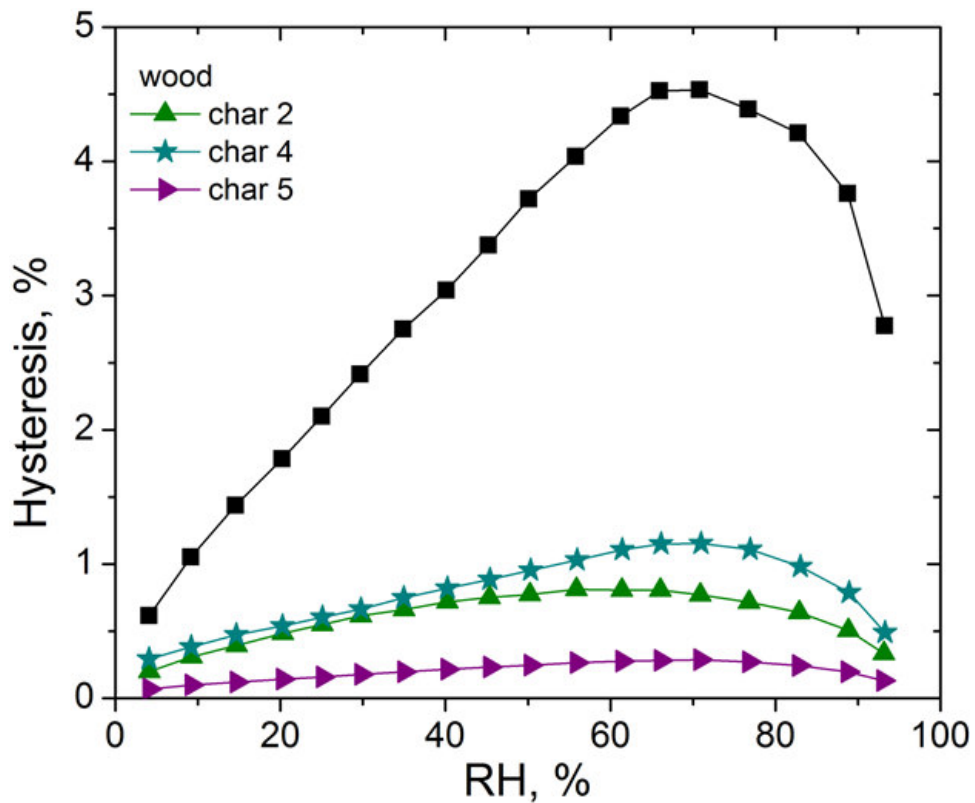


Figure 5.15. Hysteresis (%) for *Malus* wood (surface area = $0.87 \text{ m}^2 \text{ g}^{-1}$), char with low surface area (char 2, surface area = $0.85 \text{ m}^2 \text{ g}^{-1}$), medium surface area (char 4, surface area = $12.97 \text{ m}^2 \text{ g}^{-1}$) and high surface area (char 5, surface area = $37.88 \text{ m}^2 \text{ g}^{-1}$). The char was derived from *Malus* spp. under atmospheric pressure conditions and a temperature of $450 \text{ }^\circ\text{C}$ for 15 minutes.

5.4 Discussion

5.4.1 Density, surface area and pore volume

The unique properties of wood are due to the cellular structure (Bowyer *et al.*, 2002). According to Bowyer *et al.* (2002), the density is inversely related to the porosity, thereby it was expected that the density would decrease with the char samples as char is well known to be a porous material. Brewer *et al.* (2009) suggest that the presence of minerals within the char, particularly a char with high ash content, could cause a higher density as they are more dense than most forms of C. However, the mean density in this study for char was identical to the mean of the wood samples.

This study showed that the surface area increased five-fold with the charring process. This suggests that there has been a change to the composition of the cell wall properties. Kloss *et al.* (2012) found from their study that increasing the pyrolysis temperature resulted in an increase of the biochar surface area. It is thought that an increase in surface area will result in an increased number of sorption sites.

Along with variation in density across different timbers, there is considerable variation between samples of the same species, even in samples from different parts of any one tree. Variation between samples of the same species occurs due to a variety of factors such as rate of growth, site conditions and genetic composition. Systematic patterns of variation within samples from one tree are due to the position of the original sample within the tree. Generally, the more dense samples are found at the base of the tree, with density values gradually decreasing from successively higher levels in the trunk. There is also a general increase in density outwards from the pith at any given height in the trunk. The difference is marked in the rings near the pith but slows down considerably in the rings thereafter, which may be due to growth rate (Desch and Dinwoodie, 1996).

It is known that the original feedstock properties will have an effect on the biochar produced; this is due to changes within the cell wall components. The main constituents of the cell wall that are affected by the charring process are hemicellulose, cellulose and lignin (Cagnon *et al.*, 2009). These three lignocellulosic components degrade at different rates and

within distinct temperature ranges (Cagnon *et al.*, 2009). Hemicellulose degrades at approximately 200 – 300 °C, cellulose between 300 – 400 °C and the more thermostable lignin between 200 – 700 °C (González *et al.*, 2009; Kim *et al.*, 2012). During char formation at temperatures above 400 °C, the crystalline cellulose was completely broken down (Kim *et al.*, 2012). According to these figures, the biochar produced at 450 °C will have caused degradation to all three elements within the cell wall structure, though the extent to which remains unknown.

It is difficult to compare results between various char studies as all of the experimental conditions differ (Cagnon *et al.*, 2009). Pyrolysis conditions on orchards sites would also be very different to laboratory conditions as the biomass would be burnt on site, with potentially some O₂ present and also less temperature control.

5.4.2 Sorption properties of the biochar

The sorption properties of the biochar samples clearly re-enforce the notion that the charring process has had an impact of the chemical composition of the cell wall material. The decreasing adsorption/desorption with the increasing surface area of char samples are indicative of a change in the wood composition. Wood samples are more absorptive to water as the vessels are still intact, whereas these have collapsed in the charring process. Hysteresis occurs partly due to the flexibility in wood, however this flexibility is lost during the charring process. It is well known that heat treatment has an effect on a material and substantially reduces the EMC of wood (Jalaludin *et al.*, 2010a). This is due to a number of effects including the loss of hemicelluloses, cross-linking of lignin which results in a restraining effect of cell-wall expansion, and possibly a higher cellulose crystalline content (Jalaludin *et al.*, 2010a). These lead to a reduction in OH groups, due to the loss of hemicellulose and the increase in cellulose crystalline content. The OH groups are the reactive sites within the material as they are the main components attracting water and binding the water molecules by forming H bonds with them. Such binding areas are referred to as sorption sites (Engelund *et al.*, 2013). The hemicelluloses have the most sorption sites followed by cellulose and then lignin (Engelund *et al.*, 2013). These changes in the cell wall composition are again indicative of a change occurring within the wood structure upon charring. The structure of the cell wall components is also important in terms of sorption as

the wood microfibril angle determines how many sorption sites are available for water sorption (Engelund *et al.*, 2013). Any changes made to the wood material, such as through thermal modification or charring, will change the internal composition (such as the microfibril angle) and change not only the amount of sorption sites but the amount available for water sorption.

The sorption of the material is affected by the cell wall components (Jalaludin *et al.*, 2010b) and as the wood cell wall components are removed (i.e. by increased temperature or the charring process), the sorption decreases. The EMC increases with surface area as there is more area for sorption. However, in this study, the sample with the highest surface area had a decrease in EMC. This could indicate that there is a range of desirable surface area with a minimum and maximum value that is ideal for water sorption. There may be an upper limit in terms of ideal surface area for the water sorption of charred material.

Surface area seemingly has an effect on the hysteresis of the material, reiterating the fact that the wood cell wall composition may have an effect on the hysteresis of the material (Jalaludin *et al.* 2010b). In solid wood the hysteresis is caused by swelling and shrinking as moisture is adsorbed and desorbed, and it is thought that this is based on the lignin content. During the charring stage, it is believed that the components such as lignin are removed, which removes the constriction at the desorption stage as the components are able to move freely around. With the high surface area samples, the structure has broken down even more, such as the lignin within the material. There is evidence that when a substance is delignified, the hysteresis is lost. This is shown by Xie *et al.* (2010) who state that water sorption is affected by the internal geometry of the cell wall and its components. The charring process is clearly having some effect on the cell wall components, thus resulting in a change in the internal geometry.

It has been found by Harding *et al.* (1998) that porous C has a different relationship with water vapour and its adsorption. This is due to the hydrophobic nature of the C surface. González *et al.* (2009) found that the porosity of the char produced during pyrolysis is also due to the lignocellulosic composition of the feedstock biomass.

5.4.3 Implications for use of biochar as a soil amendment in UK apple orchards

Further work needs to be carried out to determine the properties of the biochar resulting from the burning of apple wood biomass on site. The pyrolysis conditions on the orchard site would differ from the laboratory conditions and research is required to determine the properties of the biochar produced and the effects that it would have on the soil when ploughed back into the system. It is important to determine whether the C locked up in the biochar is a stable store of C, which would be good in terms of sequestration, or whether it is bio-available, which may lead to problems with plant yield, due to a lack of N.

5.4.4 Further work

Many interesting questions have been raised by this study as it is apparent that charring the wood samples has had some effect on its properties. There is a very definite change, within all of the properties tested, after the charring process. The precise reasons behind these changes remain unknown but it is likely to do with changes in the wood composition after being charred. Further work should investigate what exactly happens to the wood cells and cell structure with the charring process by testing the compositional data and via microscopy. Measurements should be taken of the lignin, cellulose and hemicellulose to determine the composition within the feedstock and to determine any compositional changes upon pyrolysis. These cell wall compounds are known to have a profound impact on the structure of wood and hemicellulose is the first to degrade when wood is thermally modified. Lignin, which has the highest C content of the cell wall components, is thought to play a role in the sorption properties, although its role remains unclear. Cagnon *et al.* (2009) found that lignin is the main contributor to the final char weight, as the hemicellulose and cellulose fractions undergo higher weight loss than the lignin (76.5%, 81% and 55% respectively). Whilst determining the cell composition, it would be important to consider C:N ratios and how they may differ with the charring process. Understanding the compositional changes within the cell wall may explain the property changes when biomass is charred.

Further work should include the determination of any effects on the cation exchange capacity, which is how well the nutrients can be bound to the soil and taken up. If biochar is to be adopted by policy makers and land owners, it is important to determine the impacts

upon the nutrients within the soil. Will they be bound up in the biochar, will they be available for plant uptake, are there any implications in terms of leaching?

The comparison of the properties of biochar derived from *Malus* spp. against char made from different feedstocks would also be useful. It has been suggested that the intrinsic nature of the biomass feedstock plays an important role in the final properties of the resulting biochar (Manya, 2012). This may indicate that there is an optimal feedstock that should be used when trying to create biochar of specific properties.

The determination of the optimal charring conditions to ensure the optimal surface area availability of char. This may include studying the effect of pressure in the charcoal making process as Cetin *et al.*, (2004) found that a slight decrease of total SA by increasing the pressure. Is there an ideal pressure to form charcoal under in order to optimise surface area?

An important study would be the determination and characterisation of extractives and infiltrates in the samples to look at the spread across the log. The terpenes, resins and polyphenols such as tannins, sugars, and oils as well as inorganic compounds such as silicates, carbonates, and phosphates are locked within the cell wall, after deposition during maturation of the secondary cell wall and during heartwood formation (Bowyer *et al.*, 2002). It would be interesting to determine and quantify the effect (if any) these compounds have on the properties when charring the *Malus* wood. It would be of interest to determine what happens to these compounds in the wood during the charring process and to determine if there is any difference across the log as due to the way that the extractives and infiltrates are deposited, heartwood has a higher concentration than sapwood, therefore the density of heartwood is often slightly higher to reflect this difference (Bowyer *et al.*, 2002).

Although biochar technology offers a potential way of mitigating climate change through the storage of C in the terrestrial biosphere, it is vital that research into the longer term effects are carried out (Manya, 2012). Negative impacts of biochar must also be considered such as the release of polycyclic aromatic hydrocarbons (PAHs) from biochar. Although PAHs are released from biochar it is thought that the PAH content in wood-based biochar decreases

with increasing pyrolysis temperature (Kloss *et al.*, 2012). When considering the use of biochar as a soil amendment, Schimmelpfennig and Glaser (2012) have made some recommendations. They suggest that biochars with an O:C ratio < 0.4 and a H:C ratio < 0.6 will be the most effective in terms of C sequestration and that other standards such as the BET surface area should be > 100 m² g⁻¹.

CHAPTER SIX

EVALUATION OF ENVIRONMENTAL IMPACTS OF GREENHOUSE GAS (GHG) EMISSIONS OF APPLE PRODUCTION FROM UK APPLE ORCHARD SYSTEMS

Abstract

As atmospheric levels of greenhouse gases (GHGs) continue to rise, it is important to consider the practices which contribute to this. Whilst it is known that the food supply chain accounts for one fifth of all UK emissions, little is known about the contribution of the horticulture sector. This study investigated the GHG impact of UK apple orchards, with a particular focus on Cox and Bramley varieties; Rubens and Gala varieties were also considered to a lesser extent. Apple orchard owners completed a specific questionnaire; the data were then run through the model developed between Bangor University and Footprints 4 Food Ltd to calculate PAS 2050 compliant carbon footprints. The system boundary was cradle-to-farm gate, focussing on apple production and harvest; no post-harvest stages were considered in order to make each orchard comparable. A sensitivity analysis was carried out to explore the impact of potential management changes. The calculations showed the mean carbon footprint for Bramley, Cox and the remaining varieties (Rubens and Gala) to be 0.06, 0.11 and 0.12 kg CO₂e per kg of apples produced. Bramley orchards had significantly lower emissions ($p = 0.007$) than Cox orchards. No significant differences ($p = 0.911$) were found between the traditional and trellis planting methods or between the different tree ages; over 15 and under 15 years old ($p = 0.561$). There was moderate correlation between the carbon footprint and energy use ($r^2 = 0.33$) and a strong positive correlation between carbon footprint and fertiliser use ($r^2 = 0.75$), showing that fertiliser usage was the highest GHG contributor (ranging from 26.55 to 53.81%). In conclusion, while UK apple orchard carbon footprints are comparatively low, there is potential to lower emissions through a change in management practice.

Key words: *carbon footprint, climate change, fruit, fruit production, environmental management, Malus spp., orchard systems.*

6.1 Background and policy

6.1.1 Climate change

As global anthropogenic greenhouse gas (GHG) emissions have increased by 70% between 1970 and 2004 (IPCC, 2007a,b), there is increasing governmental and societal pressure to reduce GHG emissions in response to the changing global climate. This is especially important within agricultural and horticultural industries, with the latter being the focus of this study. Agriculture, one of the major contributing sectors to anthropogenic GHG emissions, is estimated to cause up to 32% of total global emissions (including land use change emissions) (Bell *et al.*, 2011; Bellarby *et al.*, 2008). The major causes of elevated concentrations of GHGs are the burning of fossil fuels and agricultural emissions, however, it is still important to consider carbon (C) fluxes from soils in response to changing temperatures, agricultural land management and land-use change, and the subsequent contribution to either increasing or decreasing atmospheric carbon dioxide (CO₂) levels (Bell *et al.*, 2011). Whilst it is known that the food supply chain is responsible for approximately 22% (160 million tonnes CO₂e) per annum of all UK GHG emissions (Defra, 2009), it remains unclear what contribution the horticulture industry makes to overall figures. GHG emissions from the food and drink manufacturing sector were reduced by 9.8% to around 16.2 million tonnes in 2008 from 1990 levels; with a 19% reduction in CO₂ emissions from its direct use of fossil fuels (Defra, 2010b). Total UK domestic CO₂ emissions fell by 11% to around 535 million tonnes in 2008 (Defra, 2010b). Although crop production emits a range of GHGs, including CO₂, methane (CH₄) and nitrous oxides (N₂O), the horticulture industry still lacks a roadmap of abatement options relevant to the sector which would reduce GHG emissions and energy use.

There are a number of initiatives that have been set in place in the UK aiming to reduce GHG emissions and reach targets put in place by the Kyoto Protocol (1998), under which the UK was committed to a 12.5% reduction in GHG emissions by 2008-12. There was a higher domestic goal of cutting CO₂ emissions by 20% by 2010. The Climate Change Act (2008) aims to improve C management and transition the UK towards a low C economy through a 34% CO₂ reduction by 2020 and by at least 80% by 2050. A Climate Change Programme (CCP) has been put in place by the UK Government and devolved administrations in order to help the

UK achieve Kyoto commitments (Defra, 2006). For England, a target of reducing agricultural emissions by 3 million tonnes of CO₂ equivalent per annum against a 2008 baseline by 2018 has been set by the UK Low Carbon Transition Plan (Joint Agricultural Climate Change Task Force, 2010). Correct management and response to UK climate change is required resulting in a number of abatement options leading to a lower C future.

6.1.2 Impact of the horticulture industry

At present, production horticulture (including ornamentals and food) accounts for 3% of the UK's agricultural area (Beckenham, 2009). Orchard fruit accounts for 13.6% of the industry with 20,800 hectares, with a total of 152,900 hectares production area for fruit and vegetables in the UK (Beckenham, 2009). The UK produce 2.7 million tonnes of fruit and vegetables per annum with orchard fruit, including apples, pears, cherries and plums, accounting for 284 thousand tonnes per annum (Beckenham, 2009). There is evidence that the sector is of increasing public interest and importance, with positive and lifelong health linked to proper nutrition, such as increased consumption of fruit and vegetables (Defra, 2010c; Edwards-Jones *et al.*, 2008). Poor nutrition is a huge cost to the NHS, estimated at £6-7bn per year in 2002 and set to rise (Rayner and Scarborough, 2005). It is possible to delay an estimated 42,000 annual deaths by increasing the average daily consumption of fresh produce from 3 per day up to 5 per day (Defra, 2008). Approximately two thirds of the deaths worldwide are attributed to non-communicable diseases such as diabetes, cancers, chronic respiratory diseases and cardiovascular diseases. The associated benefits of increasing consumption of fresh fruit and vegetables, along with increasing public awareness and availability, stricter regulation and a reduction in tobacco and alcohol intake, reduce the major risk factors of these diseases, thus reducing risk of premature death (Ezzati and Riboli, 2012). The UK's largest manufacturing sector is the agri-food sector, employing 3.6 million people and contributing over £80bn to the economy (Defra, 2010d). Due to the health benefits of eating fruit and vegetables there have been a number of successful campaigns to increase public awareness, such as the British Summer Fruit's Seasonal Berries campaign, Yes Peas!, Love Potatoes and Eat in Colour (Defra, 2010c). A government commitment to increasing public consumption has led to the 5 A Day Action Plan, implemented by the Department of Health and aiming to not only increase awareness but implement a

behaviour change (Defra, 2010c). An example of such a scheme promoting the increased consumption of fruit and vegetables alongside a change in behaviour principles is that of Food Dudes, pioneered by Bangor University. The campaign, aiming to prevent obesity from an early age, has seen a rise in children's fruit and vegetable consumption from 60-200% (Horne *et al.*, 2011). As fruit and vegetable consumption is being promoted, it is an important sector to consider whilst looking at the GHG emissions produced by UK industry.

6.1.3 Orchard influence

There is a perception that UK orchards are in decline, but, figures produced by Defra (2010a) show that total orchard area in England and Wales is actually increasing, and has grown by 2.5% since 2007 to give a total of 16,788 hectares covered by apple trees in the UK in 2009. As growing fruit in the UK is becoming more prevalent, the range of varieties being grown commercially is becoming more diverse. In the 2009 survey, Defra showed that the single apple variety covering most of the UK orchard area is Bramley's Seedling with 1,995 hectares, which is almost a 10% decrease on the 2007 area for this variety. The apple variety, Cox, covers the second largest area in the UK and that has also shown a decrease in area since 2007 of 16%. This decline in area of UK favourite varieties is a result of newer varieties, such as Jonagold, growing in popularity.

6.2 Carbon footprinting

6.2.1 Background to carbon footprinting

As levels of GHGs are rising in the atmosphere and concerns are growing over the current climate, there is an increased interest in estimating and quantifying the amount of GHGs that are emitted from particular farm systems and food supply chains (Edwards-Jones *et al.*, 2009; Hillier *et al.*, 2009; Taylor *et al.*, 2010). This is of increasing interest considering the world population and the rising demand for food production and increased pressure on natural resources. Policy makers, firms and consumers are becomingly increasingly committed to reducing GHG emissions and as a result possible mitigation options are being explored (Brenton *et al.*, 2009). One such option is determining the GHG emissions associated with certain products and their subsequent 'carbon labelling' (Brenton *et al.*, 2009). Labelling the product with its GHG emissions provides an indication of and effectively

communicates the global warming impacts associated with a product to the stakeholders (Edwards-Jones *et al.*, 2009; Page *et al.*, 2011). A carbon label also offers an incentive for different parts of the supply chain to reduce emissions, thus making the product more commercially popular if consumers preferentially purchase products with a lower carbon footprint (Edwards-Jones *et al.*, 2008; Lakso, 2010). 'Carbon accounting' is a process which analyses such farm systems, products and chains and considers all GHGs emitted during the cycle (Carbon Trust, 2007; Wiltshire *et al.*, 2009). This 'carbon accounting' results in the 'carbon footprint' of a given product, which is the final summary of the sum of GHGs emitted from the system under analysis (Brenton *et al.*, 2009; Edwards-Jones *et al.*, 2009; Taylor *et al.*, 2010). Any defined system, such as a farm or food product, namely apple production, can have a carbon footprint calculated. The carbon footprint of a product includes the production, processing, distribution and waste disposal, thus enabling the emissions to be calculated for a whole supply chain, or it may only include certain parts of the chain, as long as that is clearly stated in the definition of the system boundary (Brenton *et al.*, 2009; Taylor *et al.*, 2010). The carbon footprints are modelled and the calculation is expressed for a defined 'functional unit', which the item of production that is being analysed, e.g. a kilogramme of apples. The carbon footprints are expressed in units of carbon dioxide equivalents (CO₂e), with the most important GHG contributors in agriculture being CO₂, N₂O and CH₄. According to the IPCC (2007b), current estimates are that 1 kg of CH₄ is equivalent to 25 kg of CO₂, and 1 kg of N₂O is equivalent to 298 kg CO₂ over a one hundred year time period. Thus, different GHGs have been normalized with respect to CO₂ and the carbon footprint is expressed in terms of CO₂ equivalents per functional unit (e.g. kg CO₂e per kg of food produced) (Edwards-Jones *et al.*, 2008).

Firms are eager to reduce emissions from their products in response to consumer demand and as a result many initiatives are seeking to develop methodologies and standards for the estimation of the carbon footprint of food and agriculture (Brenton *et al.*, 2009; Taylor *et al.*, 2010). Any method used needs to be based on a sound, well-developed and independent scientific base (Brenton *et al.*, 2009). Currently, this scientific base is small but it is expanding (Brenton *et al.*, 2009). At least 16 different methods have been developed by retailers and countries for product-level footprints, with each one considering unique sets of variables and system boundaries (Taylor *et al.*, 2010). As different methodologies are used,

it is often difficult to compare footprint values once they have been calculated, thus leading to consumer and industry confusion. In 2008, The Carbon Trust, British Standards Institution and Defra published an initiative called PAS 2050 (Publicly Available Specification 2050) which had been devised to produce a standardized methodology to measure embodied carbon footprints (Brenton *et al.*, 2009; BSI, 2008).

6.2.2 Carbon footprinting in food and agriculture

A significant contributor to global carbon emissions is the agricultural sector, due to the production and use of farm machinery, fertiliser production and addition, soil emissions, livestock emissions (CH₄) and manure management (Hillier *et al.*, 2009). Food systems have an associated carbon footprint due to carbon emissions from the production, packaging and transportation phases of the chain. These carbon footprints are part of the total carbon footprint of an individual, an organization and of a country as a whole. It is important to quantify the emissions and to calculate the footprint to determine the impact of the system on the global environment. There are a number of factors to consider including how the food is produced, how much energy has gone into the production, the energy used in harvesting and packaging of the product and the transportation of materials involved in the process. There are three methods that have been used to calculate carbon footprints of food items: modelled, aggregated and empirical (Taylor *et al.*, 2010). Modelled carbon footprints typically rely on theoretical considerations of agricultural systems rather than on data collected from those systems or individual farms, thereby not representing any variation between regions or farms. Whereas, aggregated carbon footprints are based on real farm data that, after collection, has been combined to form a national statistic, thus allowing general observations on best practice management and showing some degree of variability. Empirical carbon footprinting is another approach, which is able to quantify GHG emissions for a particular and defined farm system and/or supply chain. This method is an applied approach based on input and process data collected directly from the farmer or the stakeholder via a questionnaire and then analysed using relevant emission factors in order to estimate a carbon footprint for that particular food item (Taylor *et al.*, 2010). This method of calculating carbon footprint will be used in this study.

There is high variation in carbon footprints over a range of production systems (Table 6.1).

Table 6.1. Carbon footprint values over a range of production systems.

Category of product	Emissions (kg CO ₂ e) per kg product	System boundary	Reference
UK intensive Cox apples	0.066	All stages included except distribution/retail, in-use, and disposal	Wiltshire <i>et al.</i> , 2009
UK extensive Cox apples	0.078	All stages included except distribution/retail, in-use, and disposal	Wiltshire <i>et al.</i> , 2009
UK organic Cox apples	0.10	All stages included except distribution/retail, in-use, and disposal	Wiltshire <i>et al.</i> , 2009
New Zealand organic semi-intensive apple (800 trees per ha)	0.0189	Cradle to gate	Page <i>et al.</i> , 2011
New Zealand organic intensive apple (1250 trees per ha)	0.0213	Cradle to gate	Page <i>et al.</i> , 2011
UK Conventional oil heated tomatoes	2.3	All stages included except distribution/retail, in-use, and disposal	Wiltshire <i>et al.</i> , 2009
UK Conventional waste heated tomatoes	0.39	All stages included except distribution/retail, in-use, and disposal	Wiltshire <i>et al.</i> , 2009
Conventional UK onions	0.42	All stages included except distribution/retail, in-use, and disposal	Wiltshire <i>et al.</i> , 2009
Organic UK onions	0.59	All stages included except distribution/retail, in-use, and disposal	Wiltshire <i>et al.</i> , 2009
UK carrot	0.35	All stages included except distribution/retail, in-use, and disposal	Wiltshire <i>et al.</i> , 2009
20 farms in the Cambrian mountains	Range from 7 to 51 (per kg of liveweight)	Cradle to gate	Taylor <i>et al.</i> , 2010
Welsh lamb from an upland farm on mineral soils	8.1 to 31.7 (per kg of liveweight)	Cradle to gate	Edwards-Jones <i>et al.</i> , 2009
Welsh beef from an upland farm on mineral soils	9.7 to 38.1 (per kg of liveweight)	Cradle to gate	Edwards-Jones <i>et al.</i> , 2009
Welsh lamb from a hill farm on primarily organic soil	20.3 to 143.5 (per kg of liveweight)	Cradle to gate	Edwards-Jones <i>et al.</i> , 2009
Welsh beef from a hill farm on primarily organic soil	18.8 to 132.6 (per kg of liveweight)	Cradle to gate	Edwards-Jones <i>et al.</i> , 2009
New Zealand organic kiwi	0.0172	Cradle to gate	Page <i>et al.</i> , 2011
Danish butter in 250g wrap	9.0 and 10.0	Cradle to grave (including transport, consumer and waste phases)	Flysjö, 2011
Danish butter in 250g tub	9.3 and 9.8	Cradle to grave (including transport, consumer and waste phases)	Flysjö, 2011
Danish butter in 10g mini tub	9.2 and 14.7	Cradle to grave (including transport, consumer and waste phases)	Flysjö, 2011

Danish blend product in 250g tub	6.8 and 7.1	Cradle to grave (including transport, consumer and waste phases)	Flysjö, 2011
Danish lower fat blend product in 250g tub	5.2 and 5.5	Cradle to grave (including transport, consumer and waste phases)	Flysjö, 2011
Swedish energy corrected milk (ECM)	0.94 to 1.33	Cradle to gate	Henriksson <i>et al.</i> , 2011
New Zealand milk	1.00	Cradle to gate	Flysjö <i>et al.</i> , 2011
Swedish ECM	1.16	Cradle to gate	Flysjö <i>et al.</i> , 2011

Each carbon footprint will contain an element of variability due to biophysical variation between the farms and production years studied. There will also be differences between the climate, soil and productivity.

6.2.3 Aims and objectives

It is important to quantify GHG emissions from the food industry and the impacts upon the environment. This study aims to determine the GHG impact of UK apple orchards on the atmosphere. In determining the impact of the industry and comparing it to the emissions of other processes, it can be assessed how the production of apples in the UK affect the environment.

The project aims to provide a holistic assessment of 16 orchards from 6 growers in the UK. The GHG emissions associated with UK apple production within these farms will be quantified. Orchards were selected to represent various factors including planting regime, orchard age and apple variety with a view to providing a comprehensive GHG footprint for each of the participating orchards. Within this study, changes within management practice were explored and the impacts on the overall carbon footprint reported and key areas within each orchard identified that offer significant potential for emissions reductions per kilogramme of product. Aspects of UK apple production which contribute the highest GHGs will be determined, thus identifying the areas of production which have the biggest environmental impacts and pinpointing areas where carbon savings can be made. The outcome will ultimately aid orchard owners with on-farm management decisions and can be used as an effective marketing tool. The study will provide relevant agencies with further knowledge and understanding of GHG emissions, thus providing a platform for future work in this area, particularly within UK horticulture.

6.3 Carbon footprint methodology

This section of the study describes the main decisions during each stage of the carbon footprinting process. According to Brenton *et al.* (2009), there are four key elements in determining the carbon footprints of products or services. These are critical to ensure high quality and reliable results, and they are: distinctly setting the system boundary; the use of primary data over secondary data; the use of appropriate and ideally peer-reviewed emission factors; and treatment of land-use change (LUC). Therefore, the orchards in this study were directly involved in providing primary data to use in the model; data were obtained for agricultural inputs and agricultural practices directly from six growers, who filled in a questionnaire for the 2010 – 2011 growing season. The system boundary was clearly defined (section 6.3.5). As the IPCC (2006) set a twenty year cut-off period for inclusion of LUC emissions, LUC is not applicable in this study as the sites have been used as orchards for twenty years or more. All sites have had orchard trees growing on them for over twenty years or more, even if they have been replaced by a young orchard. The BSI PAS 2050 definition of an emission factor is the “amount of GHGs emitted, expressed as CO₂ equivalent and relative to a unit of activity” (BSI, 2008). UK accepted emission factors were used in this study, as provided by Defra (2011) or the Carbon Trust (2010). Both sources of emission factors are regularly updated and contain only peer-reviewed data sources. The emission factors from Defra are publically available and those from the Carbon Trust are available under license. The emission factors allow the total carbon emitted from a certain process to be calculated and expressed as CO₂ equivalents (CO₂e). The total carbon emission arising from a certain part of the process can be estimated if the emission factors are known for all downstream materials and processes, such as the manufacture, transport and use of a certain amount of product. The amount of carbon emitted is calculated by the simple multiplication of the amount of product used by the relevant emission factors (Brenton *et al.*, 2009). When this can be done for each stage of the process, the total carbon emissions can be calculated for the process as a whole.

6.3.1 PAS 2050 methodology

In response to increasing public awareness of the environmental impacts of goods, services and consumer products, many methodological approaches have been developed to assess lifecycle GHG emissions. There has been a call from both community and industry for an international standard to be established which provides a consistent method (BSI, 2008; Taylor *et al.*, 2010). A transparent framework is essential to provide a common basis and make individual assessments of climate change impacts comparable (BSI, 2008). It is important to be able to compare them and obtain a coherent methodology in order to avoid misleading results (Dias and Arroja, 2012). In order to achieve this, in 2008 the British Standards Institute (BSI) published the Publicly Available Specification (PAS) 2050 which provides a benchmark containing specific requirements and guidelines for determining the carbon footprint of a product, the encompassing PAS 2050 term for goods and services (BSI, 2008; Dias and Arroja, 2012). PAS 2050 is the world's first framework methodology for product carbon footprinting and it measures the GHG emissions associated with clearly defined aspects of the production, transportation, modification, storage, usage and recycling or disposing of goods and services to determine lifecycle GHG emissions (BSI, 2008; CICS, 2012). The use of standard, internationally accepted values and emission factors is a requirement of PAS 2050 (Taylor *et al.*, 2010). The use of an international standard can lead to the development of lower carbon goods and promote emissions reduction initiatives as key GHG hotspots can be identified within a supply chain (BSI, 2008; CICS, 2012). PAS 2050 allows comparisons to be made between products, dependent on the system boundary being clearly defined and allows assessment of product supply chain GHG emissions using a recognised and standardised approach (BSI, 2008).

6.3.2 The Bangor University-Footprints 4 Food horticultural carbon footprint model

The model used was developed as part of collaborative work between Bangor University and Footprints 4 Food Ltd in order to calculate PAS 2050 compliant carbon footprints for horticultural operations on a commercial basis. The version used in this study was designed specifically for use in orchard systems. The output of the model is a carbon footprint for each orchard/growing system that can be broken down to show the total GHG emissions at each stage of the crop production process. Model outputs are expressed as CO₂e per kg of apples produced.

6.3.3 Data collection

A comprehensive questionnaire (Appendix 6.1) was developed and sent to UK orchard owners/managers, primarily with Bramley and Cox production, to obtain specific data by asking for details on the structure and function of the orchard in order to gather information regarding the energy inputs and outputs with respect to apple production. Telephone interviews with participating orchard owners provided further information in most cases. The information provided was then used to calculate the carbon footprint of the system.

6.3.4 Study sites

Data for agricultural inputs and practices were obtained from six growers for a representative production year throughout England, UK (Table 6.2). The twelve month seasonal period of UK orchard activities associated with the apple production up to and including the harvest period was the growing season 2010 – 2011.

Table 6.2. Detailed site information for each carbon footprint.

Orchard	Site	Location	Variety	Orchard age	Soil type/geology	Planting type
1	1	Cambridge	Bramley	17 – 32	Heavy silt	Traditional
2	1	Cambridge	Cox	14	Heavy silt	Traditional
3	2	Kent	Bramley	16	Thanet beds	Traditional
4	2	Kent	Bramley	13	Thanet beds	Traditional
5	2	Kent	Bramley	13	Thanet beds	Traditional
6	2	Kent	Cox	26	Thanet beds	Traditional
7	2	Kent	Cox	14	Thanet beds	Traditional
8	2	Kent	Cox	6	Thanet beds	Trellis
9	3	Hampshire	Bramley	19	Upper greensand	Traditional
10	3	Hampshire	Bramley	6	Upper greensand	Trellis
11	3	Hampshire	Cox	19	Upper greensand	Traditional
12	3	Hampshire	Cox	3	Upper greensand	Trellis
13	4	Kent	Bramley	4 – 73	Clay loam	Both
14	5	Kent	Bramley	12	Headland brickearth	Trellis
15	5	Kent	Rubens	6	Headland brickearth	Trellis
16	6	Kent	Gala	6	Clay loam over sandstone	Trellis

Each grower provided information of the energy inputs and outputs associated with the production of apples, but the number of footprints calculated per site varied based on their individual crop mix and age of their trees. The emissions and subsequently, environmental impacts of individual producer practices were calculated using energy consumption data

collected. Emissions from field operations during apple production and harvest were analysed for the orchard footprint.

6.3.5 Defining the system boundary

The system boundary is defined as a “set of criteria specifying which unit process are part of a product system” (BSI, 2008). The system boundary must be clearly defined in order to fully understand the estimated carbon footprint of the system. It can include all or parts of the food chain depending on the purpose of the study. The system boundary used in this study includes emissions arising from the manufacture and distribution of orchard inputs, the use of energy on the orchard (fuels and electricity) and emissions from soils related to fertiliser use. The system boundary not only includes this systemic boundary but also a temporal boundary (i.e. one production year) and a spatial boundary (in this study, the footprint is for a particular variety on the orchard site, not the whole orchard). In systemic terms the boundary began at the planting stage (including transport of saplings) and included all processes up to the completion of harvesting. Post-harvest operations were not included as not all of the sites surveyed had the same facilities. It was difficult to get the information from farms that do as they provide the service for many farms in the local area, which made it difficult for them to separate out their own data. In order to make each footprint comparable, the decision was made to focus on apple production and harvest (i.e. to study the cradle-to-farm gate footprint alone).

6.3.6 Functional units

A functional unit is the “quantified performance of a product system for use as a reference unit” (BSI, 2008). The GHG emissions (carbon) footprints presented in this study are for one production year and can be compared between enterprises. The functional unit of the study is 1 kilogram of apples at the farm gate, therefore GHG emissions per unit product e.g. per kg apples produced, is calculated.

6.3.7 Field emissions calculated

Field emissions calculated included planting, irrigation (where relevant), fertilizer, mineral and synthetic pesticide (pest and disease management), herbicide (management of the understorey), use of sundries, machinery and the inputs of orchard residues back into the

system. Field operations were analysed for the apple carbon footprint, including detailed calculations for field emissions and energy consumption within the apple production process to reflect producers' practices. The field emissions must be multiplied by emission factors to give the overall emissions of the process.

6.3.7.1 Emission factors

The underlying process in carbon footprinting is to calculate the amount of GHG's emitted during specific processes; this is done by multiplying the amount of an input used by its specific emission factor (EF). The emissions factors represent the total GHG emissions resulting from a specific process (e.g. 1 kWh of UK grid electricity) determined by scientific analysis. These are then peer-reviewed, compiled by several organisations and made available as databases, which are updated as the studies are repeated over time and the science base grows. Within the UK, two of the most comprehensive and well-used EF libraries are those produced by Defra (2011) and The Carbon Trust (2010). In this study, the emission factors used (Table 6.3) are from the aforementioned UK EF libraries.

Table 6.3. Emission factors used in the carbon footprints (where DP is used for figures protected by confidentiality).

Action	Emission factor	Unit	Source
Diesel HGV road freight (UK average vehicle loads) rigid >7.5-17 tonne	0.46753	kg CO ₂ e per tonne km	Defra (2011)
Diesel HGV road freight (UK average vehicle loads) articulated >3.5-33 tonne	0.17673	kg CO ₂ e per tonne km	Defra (2011)
Water supply	0.34	kg CO ₂ e per m ³	Defra (2011)
UK electricity supply	0.54284	kg CO ₂ e per kWh	Defra (2011)
Waste fraction: metal production (average of the four emission factors provided)	5.12475	kg CO ₂ e per tonne	Defra (2011)
Waste fraction: metal disposal (average of the four emission factors provided)	0.0305	kg CO ₂ e per tonne	Defra (2011)
Waste fraction: average plastics production	3.179	kg CO ₂ e per tonne	Defra (2011)
Waste fraction: average plastics disposal	1.197	kg CO ₂ e per tonne	Defra (2011)
Waste fraction: wood production	0.666	kg CO ₂ e per tonne	Defra (2011)
Waste fraction: wood disposal	-0.817	kg CO ₂ e per tonne	Defra (2011)
Diesel	3.1761	Kg CO ₂ e per litre	Defra (2011)
Nitrogen fertiliser	DP	kg CO ₂ e per kg active ingredient	CarbonTrust (2010)
Phosphorus fertiliser	DP	kg CO ₂ e per kg active ingredient	CarbonTrust (2010)
Potash fertiliser	DP	kg CO ₂ e per kg active ingredient	CarbonTrust (2010)
Fungicides	DP	kg CO ₂ e per kg active ingredient	CarbonTrust (2010)
Herbicides	DP	kg CO ₂ e per kg active ingredient	CarbonTrust (2010)
Insecticides	DP	kg CO ₂ e per kg active ingredient	CarbonTrust (2010)
Pesticides	DP	kg CO ₂ e per kg active ingredient	CarbonTrust (2010)
Lime	DP	Kg CO ₂ e per kg	CarbonTrust (2010)

6.3.7.2 Transportation of saplings

Saplings are imported from mainland Europe at one, two or three years old. Calculations were made using a Defra (2011) emission factor (For Sites 1 and 4, the emission factor chosen was for diesel HGV road freight – UK average vehicle loads – rigid > 7.5 – 17 tonne and for Sites 2, 3, 5 and 6, the emission factor chosen was for diesel HGV road freight –UK average vehicle loads – articulated > 3.5 – 33 tonne lorry), the distance in km from nursery to orchard and the weight of the saplings. The weight of the saplings was based on information from UK fruit tree supplier, Frank P. Matthews Ltd (www.frankpmatthews.com). Based on this information, the assumption is made that a one year old *Malus* sapling weighs 1 kg, a two year old sapling weighs 1.5 kg and a three year old sapling weighs 2 kg. The

weight is then multiplied by the number of saplings planted in the orchard. Saplings are transported on steel pallets, which are returned with the lorry after delivery and re-used by the nursery. As the pallets are used for many other purposes, they are not included in the calculations for UK apple production. The emissions associated with their production and disposal should be calculated for a separate supply chain.

As the planting of an orchard is only done once in the lifetime of the trees, any emissions relating to the planting activities are divided by the lifetime of the trees in order to determine the emissions for a single production year.

6.3.7.3 Land-use change

IPCC (2006) states that “one-twentieth of the total emissions arising from the land use change shall be included in the GHG emissions of these products in each year over the twenty years following the change in land use.” Therefore, in this study, the land use change emissions do not need to be calculated as all of the study sites have been planted orchards for longer than twenty years. Grubbing-out is not taken into account as the process occurs once in approximately 30 years.

6.3.7.4 Irrigation

Not every site studied used irrigation for the crop; the sites which did use irrigation systems were Site 1 (varieties: Bramley and Cox), Site 3 (varieties: Cox traditional and Cox trellis), Site 5 (variety: Reubens) and Site 6 (variety: Gala). The amount of water used in the irrigation systems varied between 1.68 and 314.35 m³ ha⁻¹ yr⁻¹ (Table 6.4). In order to calculate the footprint of the irrigation, the amount of water (in cubic metres) is determined by multiplying the number of tree stems by the annual amount used per tree. The value is then multiplied by the emission factor for water supply (Table 6.3) provided by Defra (2011) in order to give total emissions from mains water. The systems are powered by either electricity or diesel and this must be included in the footprint by using the Defra (2011) UK emission factors for electricity and diesel (Table 6.3) to give the total emissions for irrigating the sites. Cox production only on Site 3 used drip irrigation powered by electricity, whereas Sites 1, 5 (Rubens production only) and 6 used drip irrigation powered by diesel.

Table 6.4. Amount of water used in the irrigation systems where applicable.

Site	Variety	H ₂ O m ³ ha ⁻¹ yr ⁻¹
1	Bramley	1.68
1	Cox	1.68
3	Cox Traditional	21.18
3	Cox Trellis	314.35
5	Rubens	170.00
6	Gala	128.00

6.3.7.5 Use of fertilisers

Different orchards will have different fertiliser inputs in different locations due to differences in the soil condition. A range of mineral fertilisers are used in UK orchards and agrochemical emissions factors from the Carbon Trust were used to determine the total emissions for the amount of active ingredient used (Table 6.3). As data do not exist for nutrients, the emission factor for super triplephosphate was used. Where fertilisers contained more than one active ingredient, the total was split into N:P:K, then multiplied by the Carbon Trust emission factors for each individual element and then were added all together to give total global warming potential (GWP) for the fertiliser. See Appendix 6.2 for all of the fertilisers used at each site and the total amount used.

6.3.7.6 Pest and disease management

In UK orchards, both mineral and synthetic pesticides are used to control pathogens and the emissions were calculated as part of the carbon footprint. See Appendix 6.3 for all of the pesticides used at each site and the total amount used. Emission factors from the Carbon Trust (2010) were used (Table 6.3) to determine the total emissions for the amount of active ingredient used for fungicides, insecticides, and pesticides. As an emission factor does not exist for adjuvants or growth regulators, the emission factor for general pesticides was used to calculate these emissions.

6.3.7.7 Understorey management – herbicide emissions

For this study, the herbicide emissions were calculated with the Carbon Trust (2010) emission factor (Table 6.3). The sward in the orchard is maintained by herbicide application (usually twice per season). The amount of herbicide applied to the study sites ranged

between 2.5 and 13.3 litres ha¹ yr⁻¹. See Appendix 6.4 for details of the full amount of herbicide applied.

6.3.7.8 Pruning and residues

Orchard residues should be considered in the footprint as they are a source of N back into the soil, therefore a contributory factor in soil N₂O emissions. The study sites were all pruned by hand, with the residues being left on the orchard ground. Residue amounts included were based on fresh weight figures obtained from orchard owners for the amount of prunings, leaves and apples that fall to the orchard floor annually. Based on information from the orchard owners, the assumption is made that the pruned branches account for 63% of the residues, fallen leaves 27% and rotten apples fallen to the floor 10%. As the residues are all left on the orchard ground to rot and return to the system, the nitrogen (N) content must be calculated and inputted into the model. Firstly, the fresh weight was converted to dry weight using *Malus* moisture content data collected during moisture content experiments for apple and leaf matter, mean yield 85% and 70% respectively. Moisture loss was determined for shop bought Cox, Bramley and Braeburn by taking two slices, each weighing between 15-25g, cut from opposite sides of each fruit to allow for variation within the fruit, using two longitudinal cuts. The slices were then dried at 60°C for 7 days. A low temperature was used in order to avoid the volatilisation of sugars, caramelisation of the fruit and a subsequent change in chemical composition. Bramley and Cox leaves were collected from Site 7 (Chapter 2), then dried at 40°C for 7 days. The moisture content of *Malus* wood is 77% (Hoadley, 1981). Total nitrogen (N) content of the residues was then determined using published data on % N of wood, leaf pre-abscission and apple fruit. The total % N for the prunings was 0.59% and determined for the top half of 2 year old young apple wood by Roberts (1927). Hennerty and Morgan (1977) determined that the % N of pre-abscission leaves is 1.5%. The % N used for the apple fruit was 0.68%, an average taken from the figures determined by Hulme (1948) for the apple pulp, peel, fruit and seed. For this study, it was assumed (based on information provided by the orchard owners) that for traditionally planted orchards, the fresh weight of orchard residues is 13, 000 kg ha⁻¹. The annual amount of N in the residues was determined by:

Wood – leaves – apples (63% - 27% - 10%)
Fresh weight 8190.00 kg – 3510.00 kg – 1300.00 kg
Dry weight 1883.70 kg – 1053.00 kg – 195.00 kg
Therefore amount N (kg):
Wood: $(0.59 \times 1883.70)/100 = 11.11$ kg
Leaves: $(1.50 \times 1053.00)/100 = 15.80$ kg
Apples: $(0.68 \times 195.00)/100 = 1.33$ kg
Add them together to give 28.23 kg N ha⁻¹

For this study, it was assumed (based on information provided by the orchard owners) that for orchards with trellis planting, the fresh weight of orchard residues is 3,500 kg ha⁻¹.

Wood – leaves – apples (63% - 27% - 10%)
Fresh weight (2205.00 kg – 945.00 kg – 350.00 kg)
Dry weight (507.15 kg – 283.50 kg – 52.50 kg)
Therefore amount N (kg):
Wood: $(0.59 \times 507.15)/100 = 2.99$ kg
Leaves: $(1.5 \times 283.50)/100 = 4.25$ kg
Apples: $(0.68 \times 52.50)/100 = 0.36$ kg
Add them together to give 7.60 kg N ha⁻¹

Not all of the orchard owners were able to provide information on the quantity of residues per year. Orchard residues include leaves that fall off the trees, apples that fall from the trees to the orchard floor and pruned biomass material. All of the study sites left the residues on the orchard floor and therefore, the N produced in the residues are returned to the system as they are not removed. As this information was not available from each grower, based on the data available, it was assumed that all orchards under the age of 12 years old have a fresh weight residue of 3,500 kg ha⁻¹, while orchards over 12 years old have residues with a fresh weight of 13,000 kg ha⁻¹.

It is accepted that these figures are general but they are based on information given from the orchard owners who were able to provide figures on residue weights. The figures also reflect the fact that trees under twelve years old are smaller than trees over twelve years

old, due to different planting regimes. Trees over twelve year old are more likely to be planted in the traditional method, where the trees have more space (lower planting density), thus allowing them to grow much larger, with a full canopy. Trees under twelve years old will be planted along a trellis network and although there will be a higher planting density, the trees are kept much smaller, with hardly any canopy, as the production of fruit is more important than the production of wood and leaves.

One study site (Site 4) had a large age range of trees, thus accounting for different residue inputs. It was calculated that 42% was planting of trees over twelve years old, with the remaining 58% being under twelve years old. Therefore, a weighted average calculation determined that 16.22 kg N was returned back to the system at Site 4.

In order to convert N values to N₂O emissions and subsequently CO₂ equivalents, the following IPCC (2006) Tier 1 calculation equation was used:

$$\text{CO}_2 \text{ equivalents} = (\text{N}_2\text{O}-\text{N} * 44/28) * 298$$

Where N₂O–N = annual direct N₂O–N emissions from managed soils + N₂O emissions from atmospheric deposition of N volatilised from managed soil + annual N₂O emissions from leaching and runoff (values calculated from inputting orchard N data and using IPCC (2006) data).

6.3.7.9 Thinning

Thinning is a practice employed in an orchard in order to reduce the number of fruits per tree, which results in an increased fruit size, ensures evenly distributed fruit within the tree canopy and will hinder biennial bearing. In this study, any chemical thinning agents are included in the pesticide calculations.

6.3.7.10 Harvesting

Within the study sites, the fruit is picked by hand, not mechanically harvested. Once the apples have been picked, they are placed into wooden harvest bins, which are transported

to the sorting area. The diesel used to transport the harvest bins is included in the overall orchard diesel usage.

6.3.7.11 Sundries

The proportion of the predicted useful life-time of each sundry must be used to calculate emissions from sundries used within apple production. When orchard owners could not provide information on the sundries used, assumptions based on information provided by the other growers who were able to provide information was used. It would appear standard practice within UK orchards to use one wire guard (weighing 25g each) per tree to deter damage from rabbits and four plastic ties (weighing 8g each) to secure each tree.

Emission factors were used from Defra (2011) and an average emission factor was determined for metals (by taking a mean of the EFs for aluminium cans & foil, mixed cans, scrap metal and steel cans) for the rabbit guards (Table 6.3). In order to determine the emissions from the sundry items, the number of times or years that it is used must be taken into account using the following equation:

$(\text{Number of kg that it weighs} \times \text{EF} \times \text{amount used on farm}) / (\text{number of times or years used})$

Wooden harvest bins and wooden stakes are also used on farm to collect the apples during harvest and support the apple trees respectively. However, they are not included in the calculations as PAS 2050 guidelines (2006) state that material from biogenic sources should not be included; “where a product containing carbon of biogenic origin degrades and releases CO₂ to the atmosphere, the CO₂ emissions arising from the biogenic carbon shall not be included in the assessment of emissions associated with the product...Note CO₂ emissions from products containing biogenic carbon are included in the assessment of the life cycle GHG emissions via the calculation of the weighted average carbon stored over the 100-year assessment period (see PAS 2050 5.4.3), and do not need to be included here”. Therefore, in this study, the emitted CO₂ should not be calculated and the stored carbon cannot be calculated as it is not proven that the wood was grown on purpose to create the wooden harvest bins or wooden stakes. In order to qualify as a carbon store, one of the conditions is that “the material containing the biogenic C is obtained from...an input that is

the result of human actions that cause its formation for the purpose of using it as an input to a process e.g. managed forestry” (PAS, 2050). The PAS 2050 guidelines (BSI, 2008) also state that inputs representing up to 1% of the total footprint can be discounted where the data are not clear, up to a maximum of 5%. Due to the multiple use of the bins and the many years that the bins and stakes are used, their production emissions become negligible compared to the overall footprint, therefore they are excluded from the calculation.

6.3.7.12 Diesel usage

The amount of diesel used in annual apple production varied from orchard to orchard depending on producer practices. Details of the total usage of diesel on site were provided by the orchard owner and the amount per orchard was subsequently calculated. The diesel used ranged from 61.2 up to 220.0 L ha⁻¹ yr⁻¹ on the sites studied. The study sites used only diesel, no LPG or petrol was used at all during orchard operations (see Table 6.3 for emission factor). Machinery data were not provided as it was easier to provide information on fuel for orchard operations and capital goods are not included under PAS 2050.

6.3.7.13 Nitrous oxide emissions

Within the carbon footprinting process, nitrogen inputs must be considered as outlined by the IPCC (2006) due to N₂O being a powerful GHG. These inputs include direct and indirect N₂O emissions from the soil due to fertiliser use, nitrogen released to the atmosphere and environment due to managing soils, leaching and run-off. Organic soils are known to release higher amounts of N₂O than mineral soils. All the study sites consist of mineral soils, as advised by the individual growers. In order to calculate the N₂O emissions, the guidelines and calculations set out by the IPCC (2006) are used. After the individual sources of N₂O have been calculated, they are added together before being converted to CO₂ equivalents. In terms of the sites considered within this study, the potential inputs applicable are those of synthetic N fertilisers, urea application and N in above-ground and below-ground crop residues.

6.3.8 Statistical analysis

A series of independent sample T-Tests were carried out to analyse the differences between each carbon footprint in terms of variety, planting method and orchard age. Statistical procedures were carried out using the statistical package SPSS PC version 20 (SPSS Inc., Chicago, USA), with $p = 0.05$ used as the upper limit for statistical significance.

6.3.9 Sensitivity analysis – modelling changes in orchard operations

As the model has generated a detailed account of the annual orchard emissions, it enabled the impact of potential management changes to be explored. These changes to current orchard operations offer opportunities for reducing farm emissions. For each footprint, changes were modelled for a reduction in pesticide use, reductions in fertiliser use and reductions in diesel use. Each management change was modelled for a 5% reduction, 10% reduction, 15% reduction and a 20% reduction whilst assuming a fixed yield. A combination of all management changes were modelled at a 5% reduction and a 10% reduction. For each orchard, the simulated changes were compared to the footprint of normal annual orchard operations.

6.4 Results

6.4.1 Between orchard variation

The results of the orchard carbon footprints are shown in Figure 6.1. Overall, the mean carbon footprint of the Bramley orchards was 0.06 kg CO₂e per kg apples produced. The Cox production had a mean carbon footprint of 0.11 kg CO₂e per kg apples produced and the mean for the remaining varieties was 0.12 kg CO₂e per kg apples produced.

At orchard level, there is some variation to note in the footprints calculated per kg apples (coefficient of variance = 46%). There is much less variation within the Bramley orchards sample (coefficient of variance = 27%). Orchard 1 has the lowest footprint with 0.037 kg CO₂e per kg apples produced while orchard 5 has the highest footprint with 0.078 kg CO₂e per kg apples produced.

The footprint data for the Cox orchards sampled are moderately variable (coefficient of variation = 36%). Orchard 8 has the lowest footprint with 0.073 kg CO₂e per kg apples produced while orchard 7 has the highest with 0.182 kg CO₂e per kg apples produced.

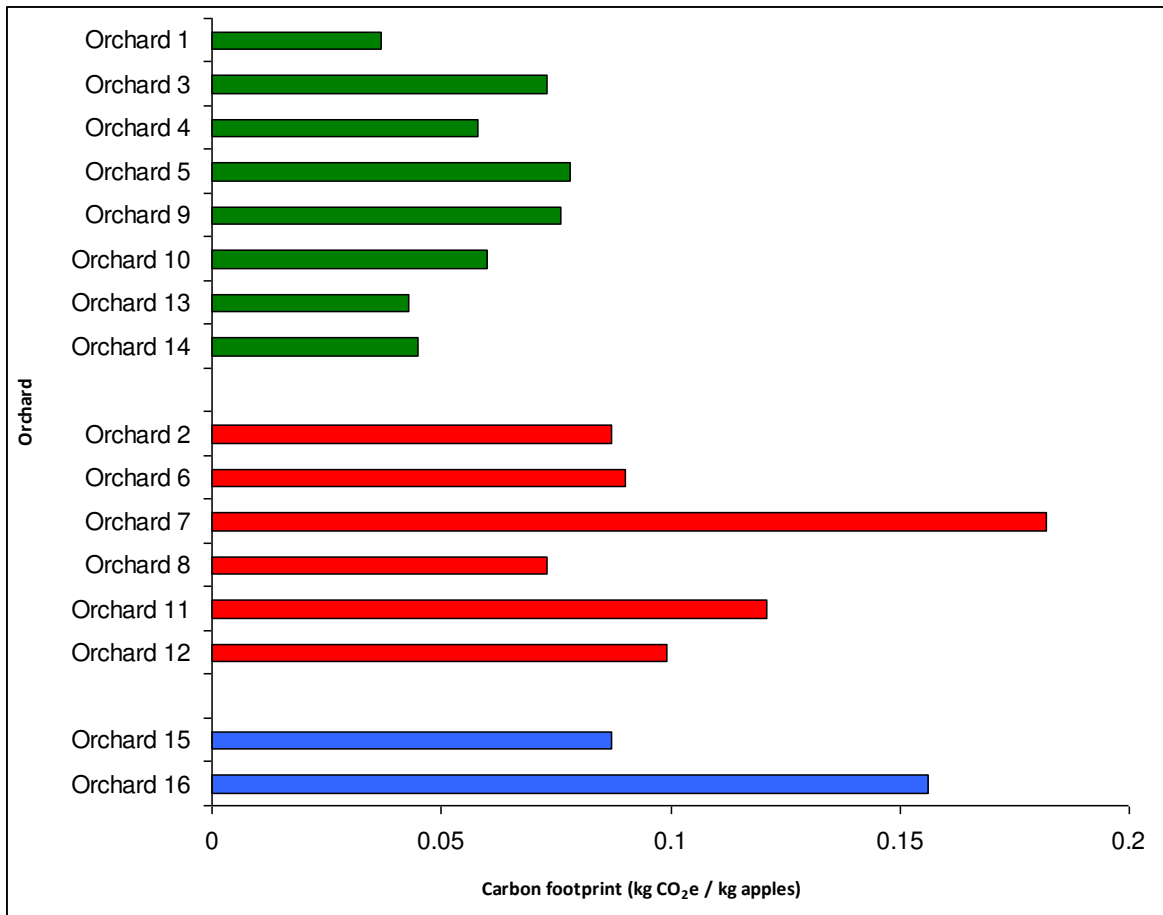


Figure 6.1. Overall PAS 2050-compliant carbon footprint results (expressed as kg CO₂e/kg apples) for each orchard in the study where green represents Bramley, red represents Cox and blue are the other varieties calculated (orchard 15 = Rubens, orchard 16 = Gala).

The analysis using an independent samples T-test (approximate normality; $p = 0.746$, and equal variance; $F = 3.061$, $p = 0.106$) show that there is a significant difference between the Cox and Bramley orchards ($t = -3.286$, $df = 12$, $p = 0.007$). The mean carbon footprint for Bramley orchards sampled is much lower (0.06 kg CO₂e per kg apples produced) than the mean footprint of the Cox orchards sampled (0.11 kg CO₂e per kg apples produced). Figure 6.2 demonstrates the difference and shows that although the mean carbon footprint of Bramley is different from the mean carbon footprint of the other varieties sampled, the average for Cox orchards overlaps with the mean for the other varieties (0.12 kg CO₂e per kg apples produced).

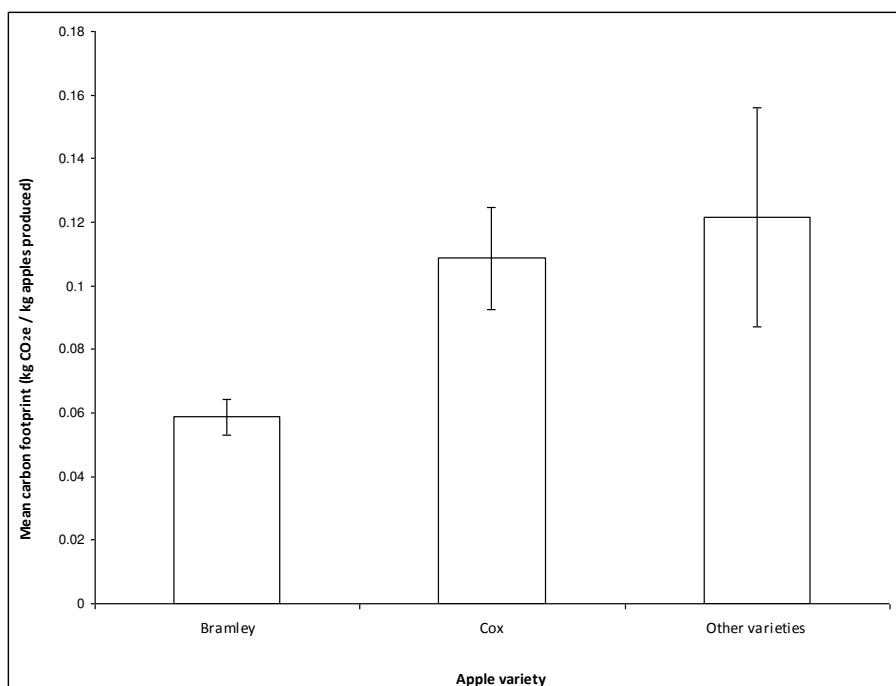


Figure 6.2. The mean carbon footprint (kg CO₂e per kg apples produced) of all the orchards sampled grouped by apple variety; Bramley (n = 8), Cox (n = 6) and other varieties (n = 2). Error bars show the standard error of the footprints within that group.

The analysis using an independent samples T-test (approximate normality; $p = 0.746$, and equal variance; $F = 3.061$, $p = 0.106$) show that there is no significant difference found between the planting methods tested ($t = 0.114$, $df = 13$, $p = 0.911$). Figure 6.3 demonstrates that the mean carbon footprint for traditionally planted orchards sampled is the same (0.09 kg CO₂e per kg apples produced) as the mean footprint of the trellis planted orchards sampled (0.09 kg CO₂e per kg apples produced).

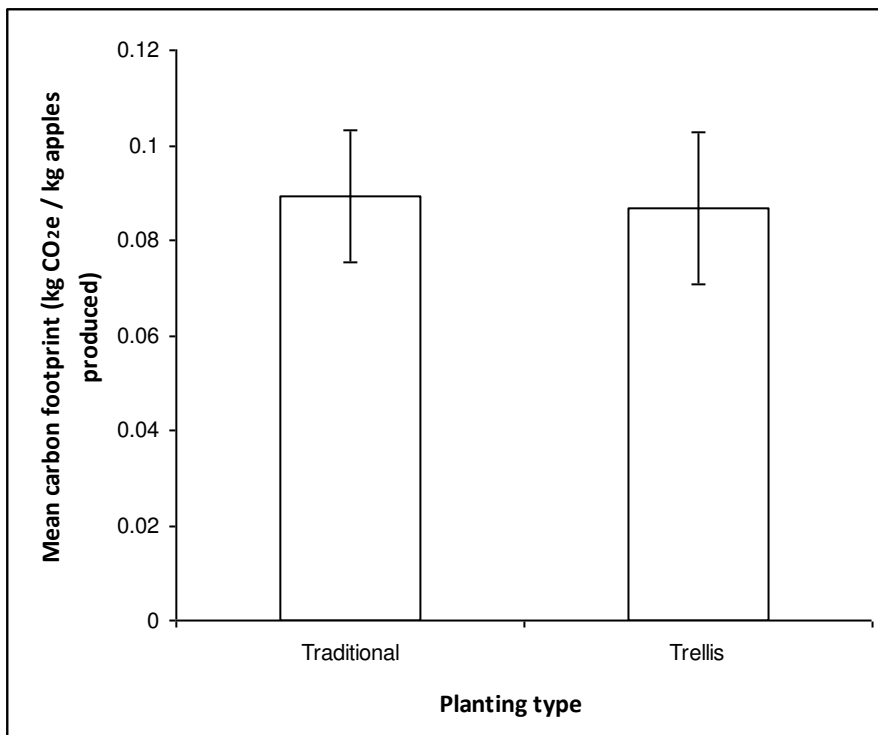


Figure 6.3. The mean carbon footprint (kg CO₂e per kg apples produced) of all the orchards sampled grouped by planting type, either traditional style of planting (n = 9) or the more modern planting on a trellis (n = 6). Orchard 13 was excluded from this analysis as it consists of both types of planting. Error bars show the standard error of the footprints within that group.

The analysis using an independent samples T-test (approximate normality; $p = 0.746$, and equal variance; $F = 3.061$, $p = 0.106$) show that there is no significant difference found between the different ages of the orchards tested ($t = 0.596$, $df = 13$, $p = 0.561$). The mean carbon footprint for young (<15 years old) orchards sampled is 0.09 kg CO₂e per kg apples produced and the mean footprint of the older (>15 years old) orchards sampled is 0.08 kg CO₂e per kg apples produced. Figure 6.4 demonstrates that although the mean carbon footprint for the more recently planted is higher than that of the older orchards, there is a lot of variation within the data.

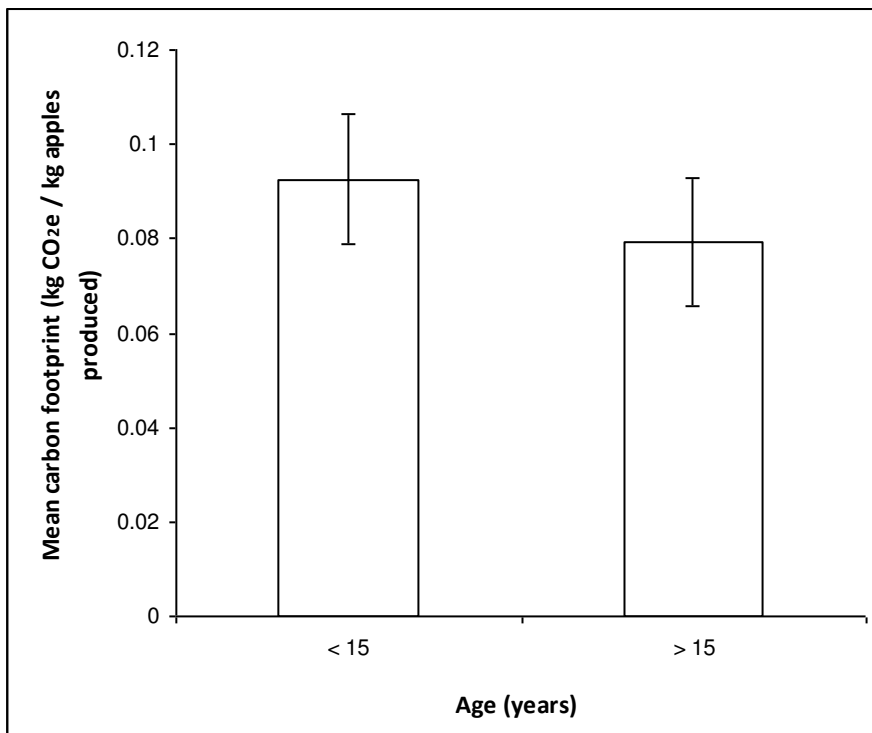


Figure 6.4. The mean carbon footprint (kg CO₂e per kg apples produced) of all the orchards sampled grouped by orchard age, with young orchards up to 15 years old (n = 10) and older orchards being older than 15 years old (n = 5). Orchard 13 was excluded from this analysis as it consists of trees ranging from the ages 4 – 73. Error bars show the standard error of the footprints within that group.

6.4.2 Footprint breakdown

The sources of emissions are shown in Figure 6.5, which represents the carbon footprint per kg of apples produced for each orchard. Using these data, it is evident that fertiliser usage is the highest contributor to the overall carbon footprint, with contributions ranging from 26.6 to 53.8%.

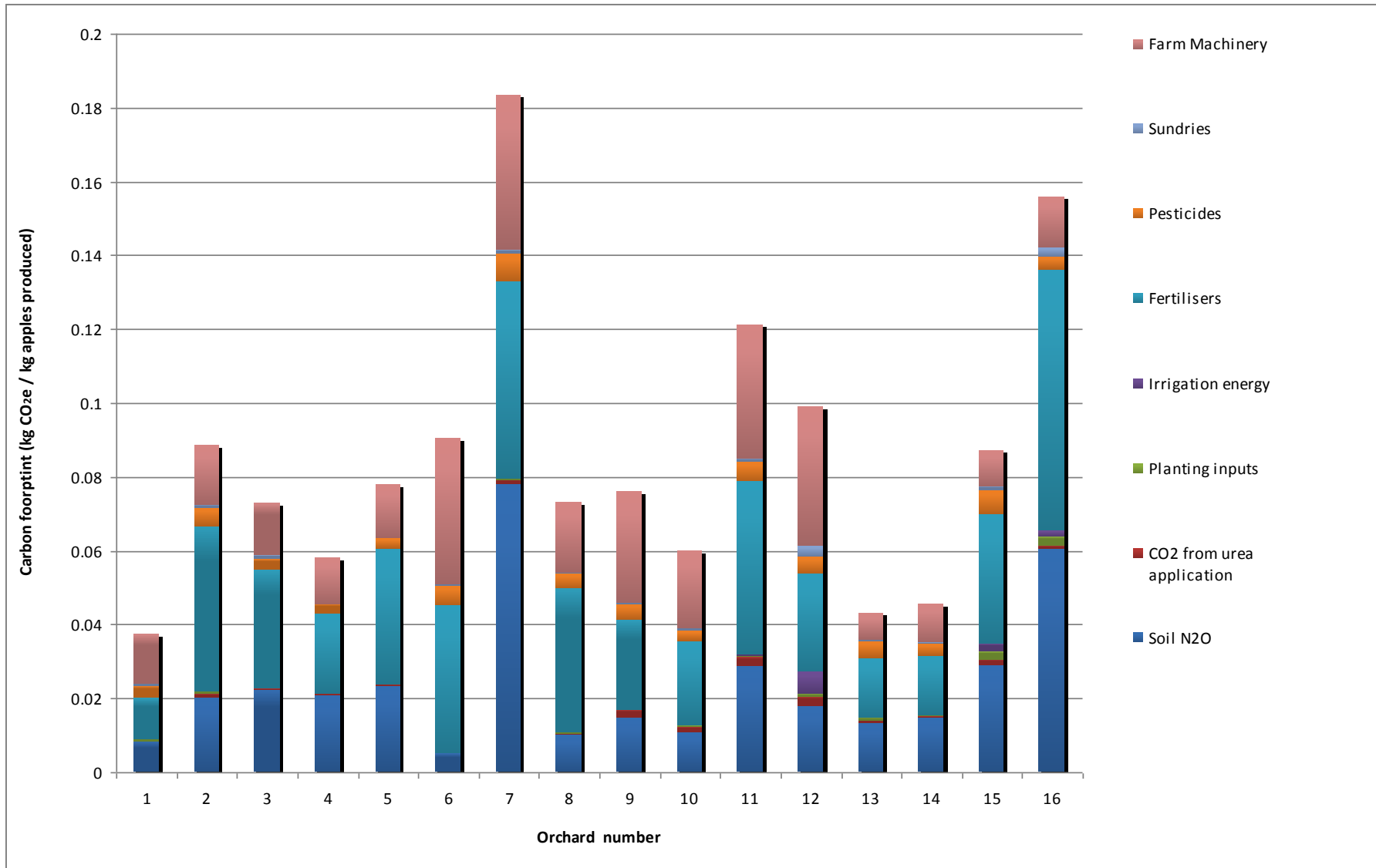


Figure 6.5. A breakdown of the carbon footprint for each orchard represented by the contribution to the carbon footprint of each management factor.

There is a moderate correlation between the carbon footprint of the orchards and energy use, which in the case of the 16 orchards studied is diesel usage as shown in Figure 6.6 ($r^2 = 0.33$).

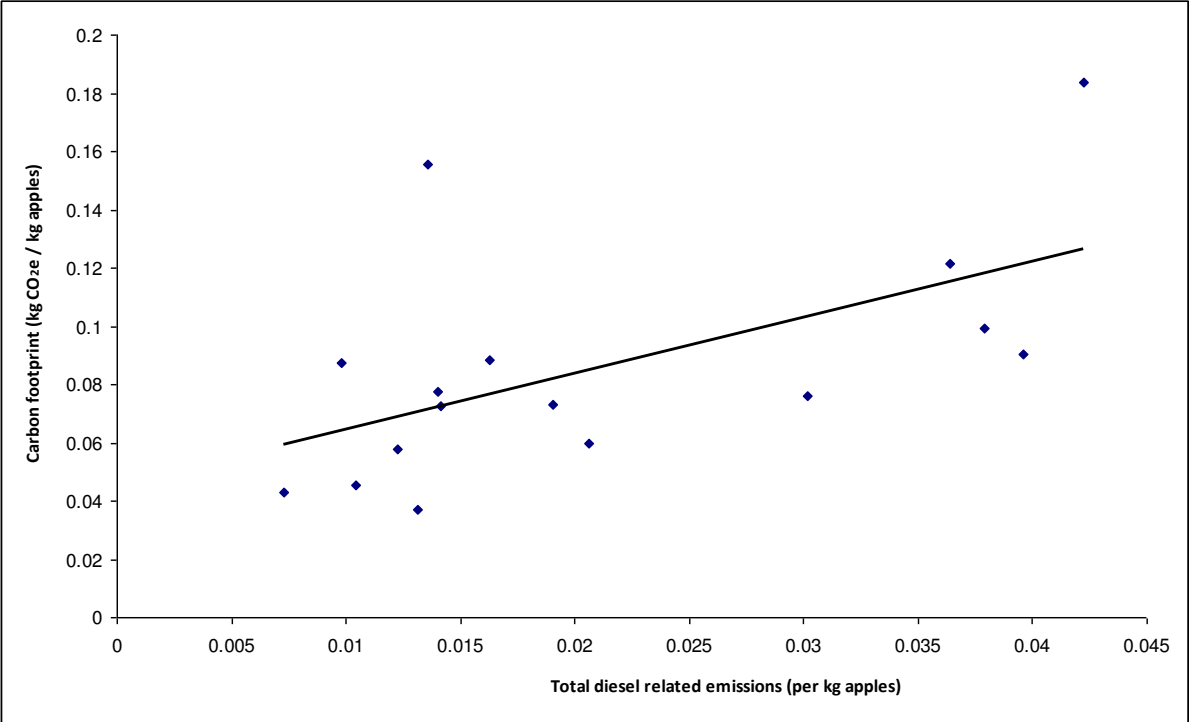


Figure 6.6. The relationship between carbon footprint and total diesel related emissions ($r^2 = 0.3334$).

Across all the orchards, emissions from the use of fertilisers contributed the most to the mean carbon footprint. Figure 6.7 demonstrates the strong positive relationship between the carbon footprint of the orchards and the emissions related to fertiliser use. These emissions include those from soil N₂O, although the soil N₂O emissions associated with crop residues have been stripped out as they are not resulting from fertiliser use ($r^2 = 0.75$).

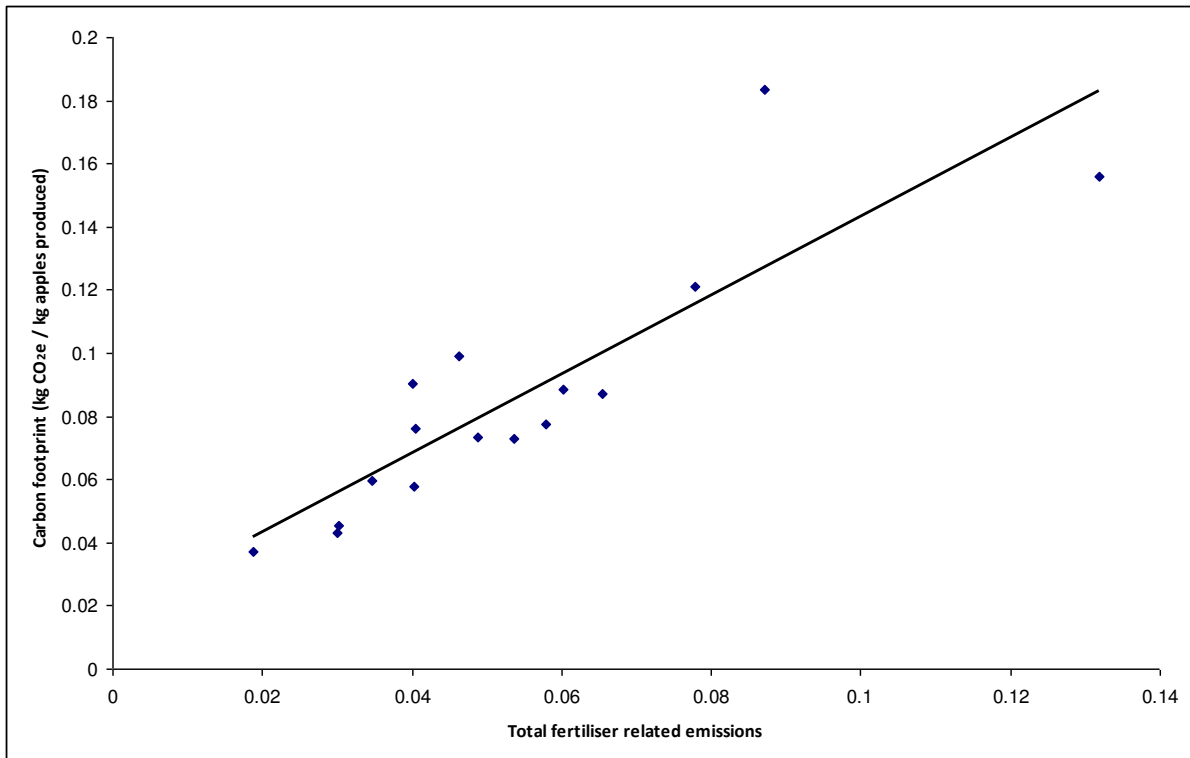


Figure 6.7. The relationship between carbon footprint and total fertiliser related emissions, including those arising from the soil N₂O (crop residue inputs have been stripped out as they are irrelevant to fertiliser usage) ($r^2 = 0.7473$). The soil N₂O emissions range from 5.49% (orchard 6) to 42.47% (orchard 7).

6.4.3 Mitigation potential

Management changes which have the potential to reduce the GHG emissions from the orchard were modelled for each of the orchards in the study, a fixed yield is assumed. Figure 6.8 shows the mean impacts of implementing the management changes. Changes in pesticide use have the smallest impact on the carbon footprint (range from 0.00% to -2.33%). Reducing the diesel usage on the orchard has a larger impact, whilst reducing fertiliser usage on site results in the largest reduction of the carbon footprint.

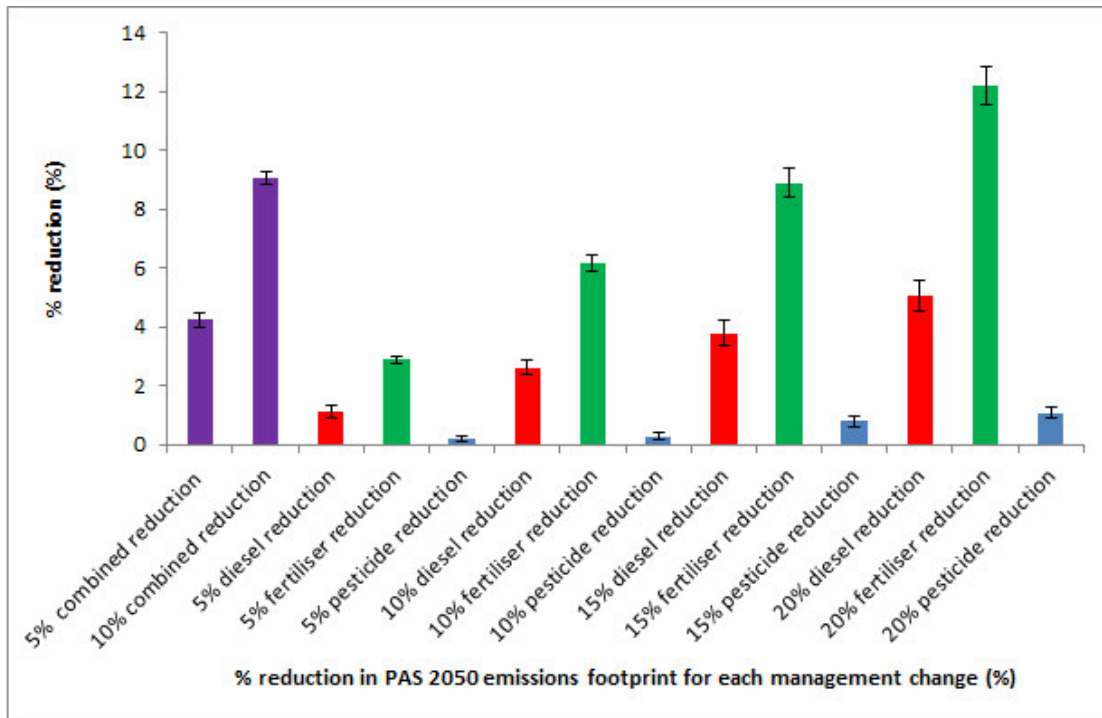


Figure 6.8. The mean impacts of implementing management changes to the orchards in the study. Reductions to diesel usage are represented in red, reductions in fertiliser usage are represented in green, reductions in pesticide usage are represented in blue and combination treatments are represented in purple.

6.5 Discussion

6.5.1 Contributing factors to the apple carbon footprints

This study estimated the empirical carbon footprints of 16 orchards in the UK, improving understanding of the GHG implications of apple production in the UK. It was shown that there were three main contributors to the carbon footprints calculated for UK apple production; fertilisers, soil N₂O and the diesel used to carry out orchard operations. Fertiliser production and use was the highest contributor to the carbon footprint in most of the orchards studied (all except orchards 1, 7, 9 and 12). The mean of the fertiliser contribution was 40.15% (ranging from the lowest of 26.55% in orchard 12 to the largest 53.81% in orchard 8). Both orchards 12 and 8 (where the lowest and highest fertiliser contributions are found) were Cox orchards growing on a trellis network of a similar age (3 year old and 6 year old trees respectively). This implies that there may be room for improvement and the potential to increase fertiliser use efficiency in orchard 8 if an orchard of the same variety, planting type and of similarly aged trees is considerably less. Of course, this may not be viable to implement as orchards 12 and 8 were on different soil types and it may be the case that the increased fertiliser inputs onto orchard 8 are a necessity due to nutrient deficiency within the soil.

The next highest overall contributor to the apple carbon footprints was soil N₂O, except for orchard 7 where it was the biggest contributor. The mean percentage of contribution was 26.0% (with the lowest being 5.5% in orchard 6 and the highest being 42.5% in orchard 7). Both orchards 6 and 7 were on the same site (Site 2), therefore they have the same soil type. They were both Cox orchards with the traditional method of planting. Orchard 6 consisted of 26 year old trees, while orchard 7 was 14 years old. The N₂O emissions from the soil were related to the amount of N applied to the orchard in synthetic and organic fertilisers, urea applications, N mineralisation (only applicable when Land Use Change was applicable), drainage and management of organic soil and the N in crop residues. For this study, only three were applicable; N in synthetic and organic fertilisers, urea application and the N in crop residues. Orchard 6, with the lowest % contribution in terms of soil N₂O, had 13,000 kg ha⁻¹ crop residues (equivalent to 28.23 kg N yr⁻¹) but had neither urea applications nor any synthetic or organic N fertiliser additions. While orchard 7, with the highest % soil N₂O

contribution, had the same amount of crop residues (28.23 kg N yr⁻¹) but did have N input in the form of 20 kg ha⁻¹ yr⁻¹ urea and 174 kg ha⁻¹ yr⁻¹ N fertiliser (34.5% N Nitram).

Diesel use on orchard sites was the third highest contributor to the carbon footprints, except for orchards 1, 9 and 12 where it was the largest contributor. The mean percentage of contribution was 25.5% (with the lowest being 8.7% in orchard 16 and the highest being 43.8% in orchard 6). Orchard 16 used 100 litres per ha⁻¹, while orchard 6 used 182.2 litres per ha⁻¹ during orchard operations. These orchards consisted of different varieties (Gala and Cox respectively), different planting types (trellis and traditional respectively) and consisted of different ages of tree (6 years and 26 years old respectively).

The remaining contributing factors; CO₂ from urea application, planting inputs, irrigation energy, sundries and pesticides, had a much lower impact on the final footprints. Although they did not contribute heavily to the GHG emissions from the orchard, it is important to remember that this does not mean they do not have any type of impact on the environment. Pesticides, for example, although they only contribute a mean of 5.47% (ranging from 2.26 to 10.31%) to the carbon footprint, can still have an impact on the environment and human health. It is important to remember that the carbon footprint does not account for these other issues such as eco and human toxicity or biodiversity impacts.

6.5.2 Assessment of orchards sampled

This study has calculated that the mean carbon footprint for Cox production is almost double that of Bramley production. This is due to Cox production having higher diesel usage and fertiliser application, which results in higher soil N₂O emissions. A combination of these factors results in the Cox orchards producing nearly twice the amount of GHGs than Bramley orchards do.

No significant differences were found between orchard age or planting system, which may be due to the small sample size as a larger sample size would enable more meaningful comparison. It is unlikely that orchard age and planting regime would not have an impact as different sized and aged orchard trees have different input requirements.

6.5.3 Comparison with other UK apple production carbon emissions

The results of this study were compared with the findings from the study carried out by Wiltshire *et al.* (2009). The work by Wiltshire *et al.* (2009) was carried out in order to test and use the final draft of the BSI PAS 2050, therefore the findings are PAS 2050-compliant and are comparable to the findings of this study. Even though the methodology used was the same, it is important to note that the system boundaries are different and they used modelled inputs, therefore the results in this study should be more accurate as they are based on real data. In this study, the system boundary was from cradle to farm gate (apple production and harvest, not including cool storage and packaging), whereas the system boundary in the Wiltshire *et al.* (2009) study included all the stages except distribution, retail, in-use, and disposal. The results of the two studies were consistent and provided carbon footprints for the apple variety, Cox. Wiltshire *et al.* (2009) calculated a range from 0.066 to 0.10, while this study calculated a range of 0.073 to 0.182 kg CO₂e per kg Cox apples produced (range of 0.037 to 0.078 kg CO₂e per kg Bramley apples produced). It would seem that the results were fairly similar with the results of this study, although there was a higher top carbon footprint value from this study. This is due to the fact that this study looked at real systems and resulted in more variable data. The next highest Cox carbon footprint in this study is for orchard 11 with 0.121 kg CO₂e per kg apples produced. This would make the findings for Cox production consistent with existing literature.

6.5.4 Comparison with New Zealand apple production

The results of this study were compared with the findings from a study carried out by Page *et al.* (2011). While the system boundaries in the two studies were the same, cradle to gate, the methodologies differ. Page *et al.* (2011) was not PAS 2050-compliant and modelled using STELLA® modelling software and Overseer® nutrient budget software. Page *et al.* (2011) did not give specific apple variety information so all the results from this study were considered (Bramley, Cox, Gala and Rubens). Page *et al.* (2011) calculated a carbon footprint range from 0.0189 to 0.0213 kg CO₂e per kg apples produced, while this study calculated a range from 0.037 to 0.182 kg CO₂e per kg apples produced. The results were reasonably consistent between the two studies with apple production resulting in low carbon emissions. However the carbon emissions from New Zealand production are slightly lower than the emissions from UK apple production. These findings were not consistent with a report by

McLaren *et al.* (2009). McLaren *et al.* (2009) quantified the emissions for the production of New Zealand apples through to the subsequent distribution, retail and final consumption in the UK. The findings were PAS 2050-compliant, therefore comparable to this study. The study found that Braeburn and Royal Gala produced carbon footprints of 0.9 kg CO₂e per kg apples produced, but the New Zealand orchard part contributed 5% and 7% respectively giving carbon footprints of 0.045 and 0.063 kg CO₂e per kg apples produced. For the organic production of Braeburn and Royal Gala the study calculated a footprint of 1.0 kg CO₂e per kg apples produced, with the New Zealand orchard contributing 10% and 9% of the overall footprint giving carbon footprints of 0.1 and 0.09 kg CO₂e per kg apples produced respectively (McLaren *et al.*, 2009).

6.5.5 Carbon emissions from apple production in comparison to other food products

The emissions related to apple production shown in this study and those carried out by Page *et al.* (2011) and Wiltshire *et al.* (2009) are low in comparison to the production of other foods. In comparison to another fruit, the kiwi, the results were similar as the carbon footprint was 0.0172 kg CO₂e per kg kiwi produced. Other food crops grown in the UK have slightly higher carbon footprints than apple production but are still low (Wiltshire *et al.*, 2009). The PAS 2050-compliant footprints range from 0.35 (UK carrots) to 2.3 (UK conventional oil heated tomatoes) kg CO₂e per kg produce with all stages included in the calculation apart from distribution, retail, in-use and disposal (Wiltshire *et al.*, 2009). Milk production also resulted in a higher carbon footprint than apple production but were still quite low, with a range from 0.94 to 1.33 L CO₂e (Flysjö *et al.*, 2011; Henriksson *et al.*, 2011). It is important to note that a different methodology was used to our study, with a standardised method of LCA using LCA software tool SimaPro 7, not PAS 2050-compliant. The carbon footprints for Danish butter and blend products were much higher than those for apple production, ranging from 5.2 to 14.7 kg CO₂e (Flysjö, 2011). It is important to note that these results are not easily comparable to our study as they are not PAS 2050-compliant.

The results of the apple production carbon footprints were compared to those of meat production, both beef and lamb from studies carried out by Edwards-Jones *et al.* (2009) and Taylor *et al.* (2010). Both of these studies are comparable to those of the apple production

as they are PAS 2050-compliant with a cradle to gate system boundary. Taylor *et al.* (2010) found a range from 7 to 51 kg CO₂e per kg of liveweight. Edwards-Jones *et al.* (2009) calculated a range from 8.1 to 143.5 kg CO₂e per kg of liveweight for Welsh lamb and a range from 9.7 to 132.6 kg CO₂e per kg of liveweight for Welsh beef. Carbon emissions from apple production were considerably less than those of meat production, thus having a smaller environmental impact.

6.5.6 Implications of the findings

Although the mean carbon footprint for apple production in the UK has been shown as relatively low in comparison to other UK food systems, there is room for improvement within UK orchards to reduce their GHGs emissions resulting in lower cradle to farm-gate carbon footprints. Improvements focus on changes in farm management such as the timing of field operations and identifying potential emission reductions. The identification of these carbon saving areas will aid on-farm management decisions and may lead to changes in management practice. There should be a focus on maximising N-fertiliser efficiency without impacting yield, which should be of financial benefit to the orchard owners by reducing fertiliser costs and diesel usage. Options that should be considered include the efficient mechanisation by the annual calibration and maintenance of spreader equipment and minimising spray before rain events to increase fertiliser efficiency. There is also room for improvement for the orchards which use a high amount of diesel per hectare (for example, orchard 6 whose carbon footprint comprises 43.79% diesel usage) to reduce the level of diesel used in line with other orchards (the carbon footprint for orchard 16 comprises only 8.71% diesel usage).

The results of this study showed that the carbon footprint of UK apple production was relatively low in comparison to the production of other UK products. UK grown apples are truly a component of a sustainable diet, they are healthy and low carbon. Although this is a positive outcome for orchard growers in terms of climate change and their subsequent effect on the environment, it is important to remember that people require a balanced diet. It is interesting to make comparisons of apple production with the production of other foodstuffs, but it is not suggesting that people replace meat with apples in their diet. People must eat a balanced diet, including carbohydrates, protein, dairy, fats and fruit and

vegetables. In terms of reducing carbon footprints and ultimately GHG emissions, it would be interesting to investigate the possibility of changing societal perception. Can people be encouraged to eat more low GHG products, for example, chicken and pork to replace lamb and beef, which are high GHG products. Perhaps it would be of interest to determine the carbon footprints at a different level. As 1kg of meat is far more calorific than 1kg of apples, potentially instead of calculating the emissions per kg of produce, it would be interesting to think in terms of emissions per Kcal or even per nutrient.

6.5.7 Limitations to the study

There are limitations to the study in terms of sample size, a larger number of UK apple orchards studied would give enable a more meaningful and robust comparison. It would also be beneficial to compare the UK apple production with carbon footprints of other UK tree fruit production, such as pears and cherries.

Throughout this study, assumptions had to be made when farmers could not provide information. This approach is better than having no information at all but the study would be more robust with actual values for each of the orchards studied to remove any chance of error in the calculation.

This study did not calculate GHG emissions for any processes beyond the harvest stage. Although they are comparable to each other and therefore the results are valid to draw comparisons, it would be of use to include the post-harvest stages and determine the impact of the entire supply chain. Undoubtedly, sorting, cleaning, cool storage, packaging and transport through to the retail stage of the apple, will have an impact on the carbon footprint. This further calculation would enable a more accurate view of the GHG emissions of UK apple production. These added stages will play a significant role in affecting the Shop-gate footprints and provide a more balanced view on the GHG emissions of home-grown produce versus imported produce.

6.5.8 Future work

6.5.8.1 Retail trade offs

The calculation of a more complete UK apple carbon footprint, which included post-harvest information, would allow for the investigation of retail trade-offs. Further research is needed to make comparisons between the GHG emissions of imported apples against UK grown produce. Imported apples will have large transportation GHG emissions, whereas home-produced apples will not have the same large transportation GHG emissions, but will have emissions involved with cold-storage in the UK for six to twelve months until the produce can be sold the following Spring/Summer. The season, therefore, has a high impact on the benefits of apple production, an apple produced and eaten during the autumn in the UK, uses less energy than one produced in the same orchard but eaten during the following August (Edwards-Jones *et al.*, 2008). With this further research, decisions could be made by policy makers and retail managers on the most ideal balance of imported and home-grown produce to be made available in the UK, considering both environmentally and economically viable options.

6.5.8.2 Determination of optimum management practices to maintain yield

It is important to investigate the effects of mitigation measures on tree productivity as currently a fixed yield is assumed when modelling management changes but this may not be the reality and a balance needs to be found between reducing GHG emissions and producing maximum yield to ensure economic viability.

6.5.8.3 Biochar process

As further research on the addition of biochar as a soil amendment is carried out, it is important to also consider the GHG emissions associated with this process. However, as it is a separate process, it has not been included in this study.

6.5.8.4 Carbon sequestration

In order to determine a complete view of the carbon footprint on the orchard as a whole, it would be valuable to consider the levels of carbon sequestration on each site. An additional survey could be carried out to gather detailed information on tree cover (both apple trees and other species used as windbreak/hedgerows), soil, habitat extent and management and

then carbon sequestration can be calculated by estimating growth rates for trees of different species on site. The various components considered include above- and below-ground biomass, deadwood and litter, and soil, and calculations include the species mix, age and planting density for each stand of trees. The carbon sequestration value can then be converted and used to offset the orchard GHG emissions. Taylor *et al.* (2010) found that farms with the highest levels of carbon sequestration had high sequestration levels in woodlands and isolated trees, thus emphasising the high potential of trees to sequester carbon. There is uncertainty with sequestration associated with woodlands, due to the final use of the timber. If the wood is used to construct items, the carbon remains locked up in the item. However, if it is burnt at the end of the rotation, the carbon will be released into the atmosphere. This can still have a positive effect on the atmosphere in terms of GHG emissions reduction if the timber burning is instead of fossil fuel use (Taylor *et al.*, 2010). Currently under PAS 2050 guidelines this offset cannot actually be used as biogenic carbon stored in vegetation and soils cannot be assumed to “remain[s] removed from the atmosphere for one year or more following production of the product” or to remain stored in a construction product for a defined proportion of the 100 year CO₂e measuring period. It does however, allow for an interesting comparison.

CHAPTER SEVEN

7.1 General discussion and conclusion

This study aimed to determine the carbon (C) storage potential of UK apple orchards with particular focus on the most widely planted commercial varieties Bramley and Cox (Defra, 2007; English Apples and Pears, 2012). Apple production within the UK had declined by 3% in 2007 from 2004 figures, but orchard area had increased by 1.7% in 2009 (Defra, 2010a). According to research carried out by the Horticultural Development Company (Beckenham, 2009), there was a total of 152,900 ha of fruit and vegetable production area with 13.6% of that being commercial orchards, which produce 284,000 tonnes of orchard fruit per annum. While UK fruit self-sufficiency, number of registered apple growers and production area have steadily declined in recent years, the volume produced has actually increased due to improved yields, new technology and a more efficient planting method (Beckenham, 2009; Borrie and Potter, 2005). Throughout the study, it has been shown that the longer apple trees are allowed to stand and accrue biomass, the higher the C storage will be. However, in a world where efficiency and being cost-effective are the driving forces of industry, orchard trees are grubbed-out at the end of their productive life. The emphasis should be on working alongside orchard owners to help them achieve the highest possible yields in the most C saving efficient manner and quantifying the C inputs and outputs to determine just how much C is being stored within UK orchards. In the current climate, it is becoming more important to encourage orchard owners in producing high quality fruit yields with the UK government sanctioning schemes encouraging a healthier lifestyle and diet by increasing the consumption of fresh fruit and vegetables (Beckenham, 2009; Food Standards Agency, 2006; Fresh Produce Consortium, 2008). There is a drive to boost UK industry, with particular groups promoting local produce, such as English Apple and Pears (2012) who represent England at the World Apple and Pear Association (WAPA; 2012).

Chapter Two of this study quantified the C in UK orchards by determining soil C, above- and below-ground biomass, determining the amount of C stored over a 100 cm deep soil profile for one UK orchard and comparing the top 5 cm soil C to that of surrounding land uses. The work showed that orchards have a greater potential for C storage than other land uses such as arable farming but that they store less C than woodlands do. Orchard fields are subjected

to much less disturbance (approximately once every 15-35 years) than arable crop areas, which undergo annual ploughing. The C stored within the soil and biomass of the orchards varied depending on their age category, which is an indicator of management practice. For orchards older than 15 years old, they represent the more traditional method of planting, with a lower number of trees per ha that are allowed to accumulate a high amount of woody biomass. A new planting method is seen in the trees under 15 years old, where there is a vast increase in stand density and the trees are grown along a trellis network, ensuring that most C is directed to fruit production and not woody biomass. The orchards > 15 years old had significantly more C storage than those in the < 15 category, but those younger orchards still had significantly more C stored in the top 5 cm of soil than the arable fields.

The management practice of grubbing-out whole orchards was explored (Chapter Three) to determine actual C loss from an orchard during the process. Although the total biomass of the tree is being removed (up to approximately 25 t C ha⁻¹ for the above-ground biomass and approximately 5 t C ha⁻¹ for the below-ground biomass), only 0.01 t C ha⁻¹ was calculated to leave the soil during the event. Compared to the amount of soil C that accumulates over the orchards growing period, the majority will remain sequestered in the soil and accumulate over time.

In Chapter Four, the idea was explored that the biomass removed from the orchard during the grubbing-out events could be used to produce biochar, which can be used as a C rich (approximately 80%) soil amendment to return a proportion of the biomass C back into the orchard field. In accordance with other UK studies it was found that while the biochar addition to the soil did not have any adverse effects on soil quality and tree growth, it did increase the total soil C by 25% (25 t ha⁻¹ treatment) and 102% (50 t ha⁻¹ treatment). Using biochar as a soil amendment needs further investigation with longer-term field trials to determine the full effects that it has on an ecosystem before being incorporated on a large-scale. Full environmental risk assessment must be carried out as the amendment is permanent once added into the soil.

Chapter Five further investigated the sorption properties of the *Malus*-derived biochar as they play a significant role in determining the effect that biochar will have on the soil-

nutrient interactions and the effect of biochar on herbicide and fertiliser applications (Lehmann *et al.*, 2011). The study found that the charring process had a significant effect on the surface area of the material, with the resulting biochar having significantly higher surface area due to having fewer large pores than the original apple wood. It was also determined that charring time had an effect on the physical properties of the biochar. There was a much lower equilibrium moisture content (EMC) for charred samples and the moisture adsorption isotherms displayed different characteristics. Definite physical changes occurred during the charring process upon the wood cell components, this area now needs to be explored to determine exactly which changes occur. The determination of the compositional change may lead to a greater understanding of how the biochar will interact with nutrients and other compounds within the soil ecosystem.

Carbon footprints of 16 UK apple orchards were calculated (Chapter Six) to determine the impact of producing apples in the UK upon GHG emissions. The system boundary was cradle to farm-gate and did not include post-harvest or retail stages for comparison purposes. Significant differences were again found between apple variety, with Cox having a significantly higher carbon footprint (mean = 0.11 kg CO₂e per kg apples produced) than Bramley (mean = 0.06 kg CO₂e per kg apples produced). Interestingly, there were no significant differences in terms of GHG emissions between the planting method. Orchards < 15 years old had a carbon footprint of 0.09 kg CO₂e per kg apples produced, while orchards > 15 years old were found to have a carbon footprint of 0.08 kg CO₂e per kg apples produced.

Figure 7.1 shows the C storage within UK orchards in terms of the variety (Cox and Bramley) and the two age groups studied (< 15 years old and > 15 years old). It encompasses all of the C considered within the scope of this study and shows the mean GHG emissions calculated from the carbon footprints (this is an addition to Figure 2.10).

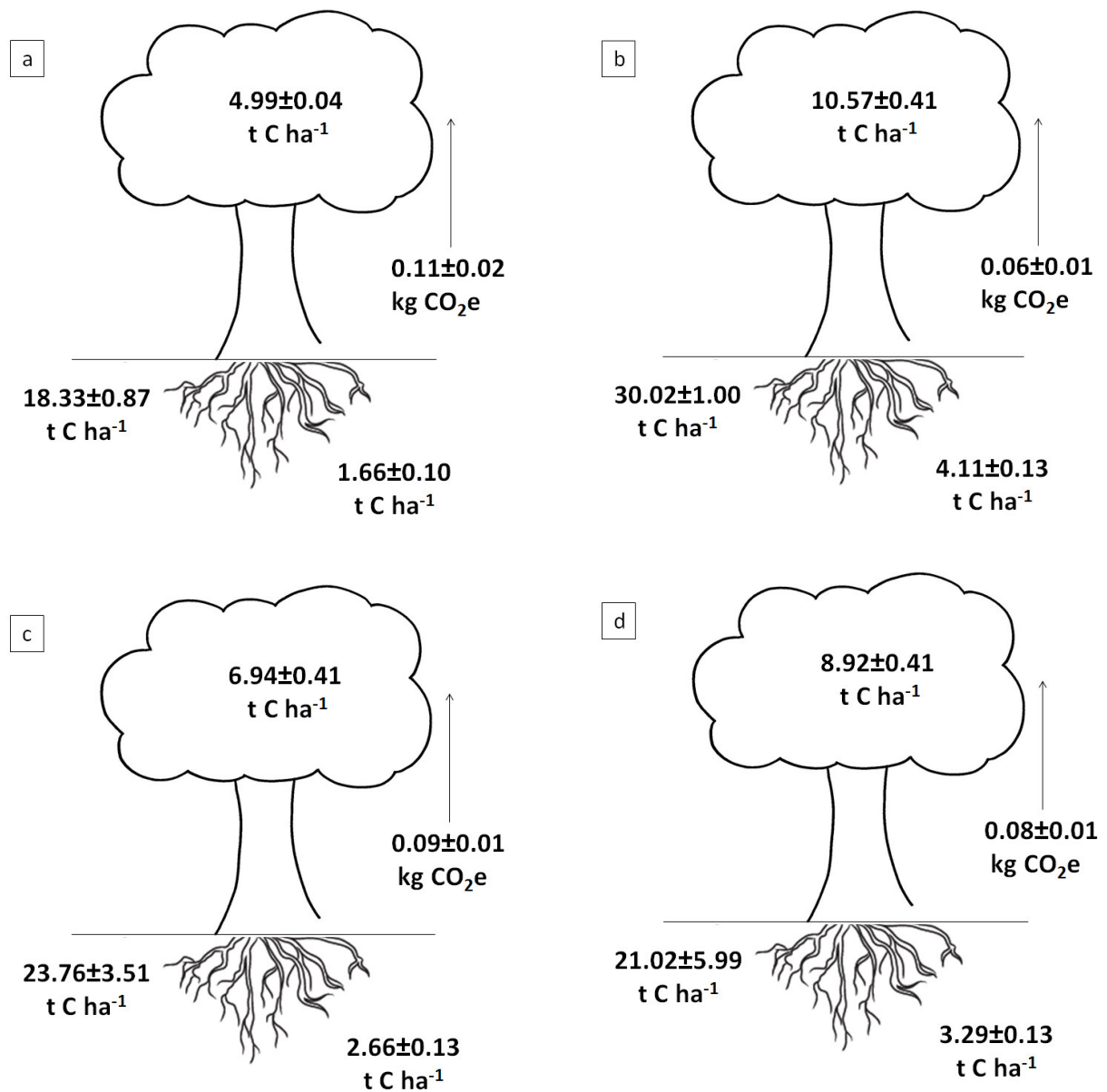


Figure 7.1. System diagrams to show the C storage within UK orchards. Figure a. shows the total C for Cox orchards, Figure b. shows the total C for Bramley orchards, Figure c. shows the total for < 15 (both Cox and Bramley) orchards and Figure d. shows the total for > 15 (both Cox and Bramley) orchards. The arrow indicates the mean carbon footprint, the value on the bottom left indicates soil C for 100 cm profile (except for Figure d. which indicates top 70 cm due to missing data values), and the value on the bottom right indicates root C. Standard error is shown for each value.

7.1.1 Future work

Future work could consider the C storage scenarios of different apple varieties as there has been a shift from the traditionally popular varieties in the UK. Bramley has declined 4% and Cox by 32% since 2004, while the area of Braeburn trees planted has increased by 40% and Gala by 11% (Defra, 2007). This study highlighted that there was a difference between Bramley and Cox in terms of C storage potential, therefore, there may be other varieties under which more C would be able to accumulate acting as a mitigation tool to climate change. Alongside quantifying the parameters of actual C storage, surveys about the popularity among retail outlets and consumers would be useful in order to find an optimum variety that would be economically viable for orchard owners to grow and be beneficial to the environment in terms of C storage, possibly being eligible for a C credit scheme should the research evidence show that the orchard is really a viable mechanism of C sustainability.

Full C budgets should be modelled using the data C in above-ground biomass, below-ground biomass, soil, carbon footprint (indicator of the GHG emissions) and parameters not covered by this thesis such as net photosynthesis and leaf respiration, fruit yield, soil respiration, root respiration and turnover and residue inputs. Microbial activity should be recorded and used to determine how much C in the residues is released as CO₂ during degradation (Lakso, 2010; Sofu *et al.*, 2005; Wibbe *et al.*, 1993). This full C budget would allow the annual C sequestration to be determined, revealing the full impact of orchard systems on the environment in terms of C sequestration in relation to other land-use types. In terms of C, it is not realistic to just look at the orchard production system alone. A full view of the system is needed with respect to processing, packaging, storage, transportation and the processes after sale, such as that in the home.

Whilst the study showed that more C is stored under larger, older apple orchards, it was realised that leaving orchards to grow into mature apple trees, such as those found in traditional cider orchards is not a viable option for commercial apple orchards. A number of factors must be taken into account, not least of all business and economics. Although all of the orchard owners who took part in the study were interested in the environment and willing to aid research (through offering their orchards for study and through the funding of

the HDC levy) into increased C sequestration, in reality the thing of utmost importance is that of maximum yield and quality of production. In the view of this, perhaps the most important aspect of future research would be into developing the work carried out in Chapter Six on carbon footprinting. Work to be done in this area includes C sequestration on site, which would give value to the trees in the orchard in terms of C storage, and quantifying the amount of C sequestration on site using the data from Chapter Two to more accurately quantify the C sequestration value of apple orchards. This “real-life” data approach may play a significant role in pushing orchards forwards towards a C credit scheme. As part of this, investigation on the effect of the grubbing-out process on the carbon footprint should be carried out and calculated within the scope of the carbon footprint. It was not included in this study as it was not necessary under current IPCC guidelines on land-use change (any land that has been under the same land-use for the past 20 years does not need to be included). However, the management practice of grubbing-out is a disturbance event to the orchard site and the effect on the full GHG emissions of the orchard should be quantified using the real on farm data for the UK collected in Chapter Three.

Further investigation into the application of biochar as a soil amendment is required. However, this study showed that it may not be a priority in terms of UK apple orchards. In reality, in the current climate, it is unlikely that orchard owners will either invest in a kiln or pay to send the feedstock to be pyrolysed. The study in Chapter Four showed that initially there seem to be no detrimental effects on orchard tree growth. Should the use of biochar as a soil addition become accepted or included in future agricultural studies, long-term field trials are required. However, currently for orchards, who may continue to incorporate small quantities on site rather than invest in large-scale production, this area of research is not the top priority. As there is such an interest in the use of biochar in this way, agricultural field research and studies into the properties of biochar (such as those in Chapter Five) should continue in a broad sense covering many systems, not just apple orchards.

CHAPTER EIGHT

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Appendices

Appendix 2.1: Soil Carbon maps

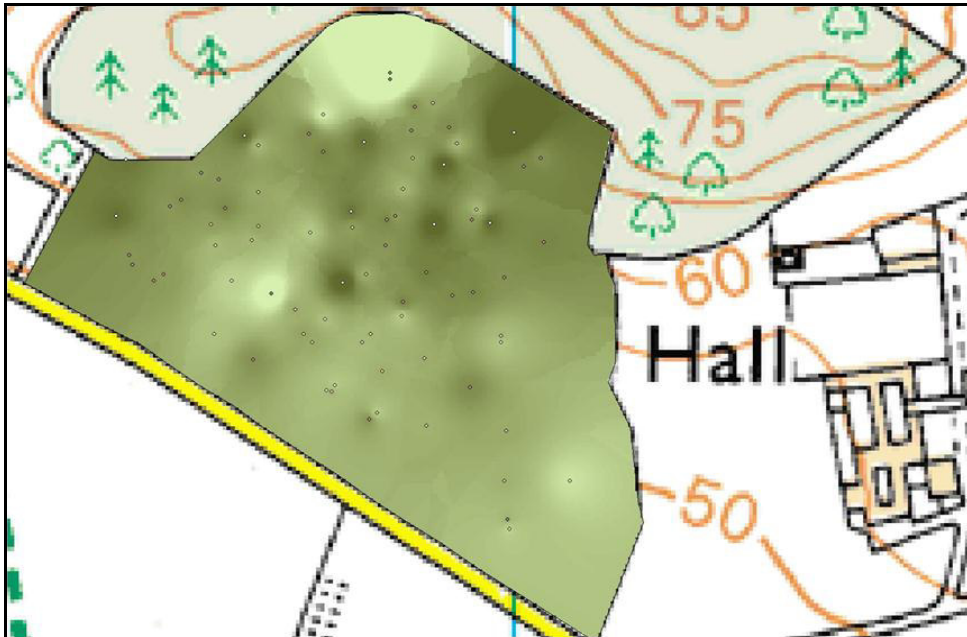


Figure 2.1.1. Site 1 map indicating soil C using data collected (the dots on the map) and interpolated to create a continuous surface using the inverse distance weighted (IDW) method on ArcMap (ArcGIS version 9.3.1 ESRI, 2009). Areas in a darker shade indicate higher levels of carbon. © Crown copyright / database right 2010. An Ordnance Survey / EDINA supplied service.

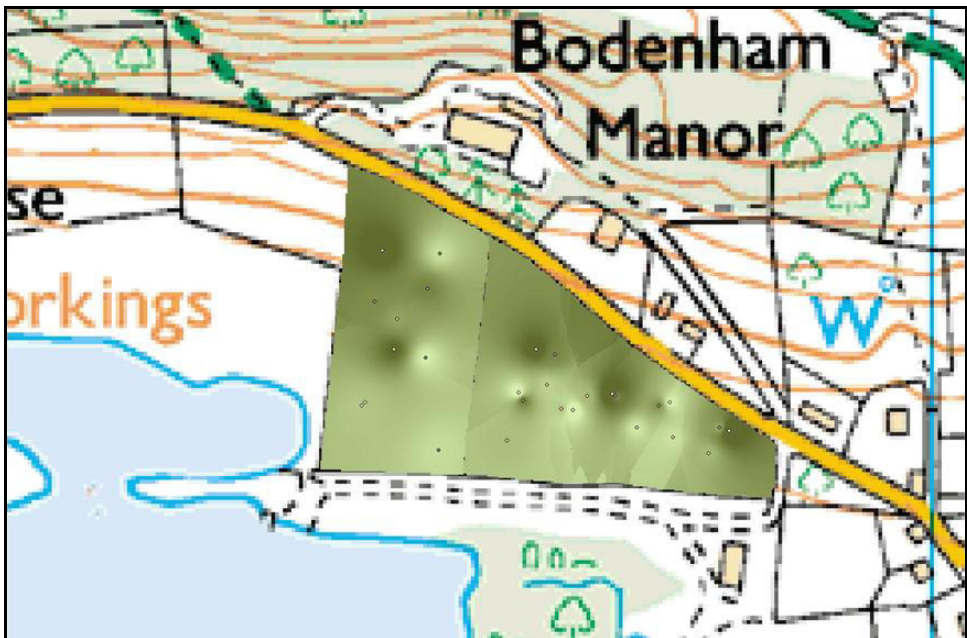


Figure 2.1.2. Site 2 map indicating soil C using data collected (the dots on the map) and interpolated to create a continuous surface using the inverse distance weighted (IDW) method on ArcMap (ArcGIS version 9.3.1 ESRI, 2009). Areas in a darker shade indicate higher levels of carbon. © Crown copyright / database right 2010. An Ordnance Survey / EDINA supplied service.

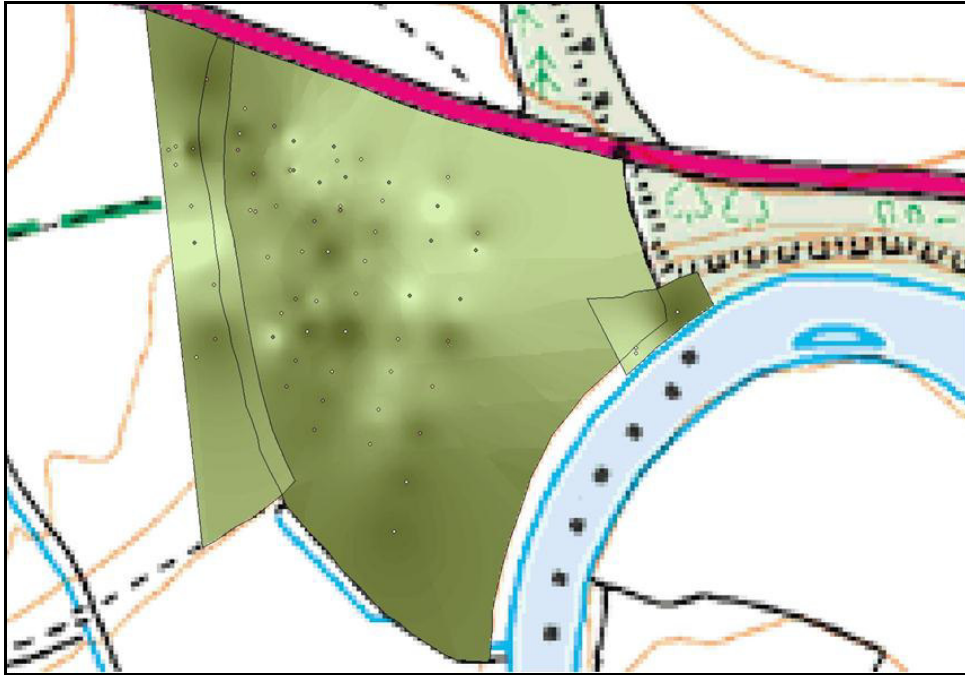


Figure 2.1.3. Site 3 map indicating soil C using data collected (the dots on the map) and interpolated to create a continuous surface using the inverse distance weighted (IDW) method on ArcMap (ArcGIS version 9.3.1 ESRI, 2009). Areas in a darker shade indicate higher levels of carbon. © Crown copyright / database right 2010. An Ordnance Survey / EDINA supplied service.



Figure 2.1.4. Site 4 map indicating soil C using data collected (the dots on the map) and interpolated to create a continuous surface using the inverse distance weighted (IDW) method on ArcMap (ArcGIS version 9.3.1 ESRI, 2009). Areas in a darker shade indicate higher levels of carbon. © Crown copyright / database right 2010. An Ordnance Survey / EDINA supplied service.



Figure 2.1.5. Site 5 map indicating soil C using data collected (the dots on the map) and interpolated to create a continuous surface using the inverse distance weighted (IDW) method on ArcMap (ArcGIS version 9.3.1 ESRI, 2009). Areas in a darker shade indicate higher levels of carbon. © Crown copyright / database right 2010. An Ordnance Survey / EDINA supplied service.



Figure 2.1.6. Site 6 map indicating soil C using data collected (the dots on the map) and interpolated to create a continuous surface using the inverse distance weighted (IDW) method on ArcMap (ArcGIS version 9.3.1 ESRI, 2009). Areas in a darker shade indicate higher levels of carbon. © Crown copyright / database right 2010. An Ordnance Survey / EDINA supplied service.

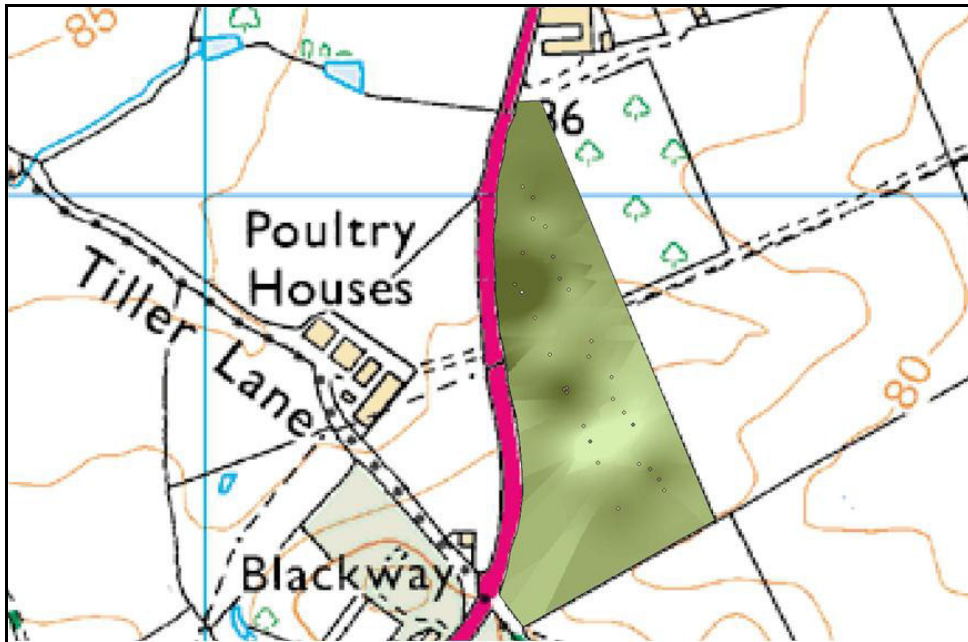


Figure 2.1.7. Site 7 map indicating soil C using data collected (the dots on the map) and interpolated to create a continuous surface using the inverse distance weighted (IDW) method on ArcMap (ArcGIS version 9.3.1 ESRI, 2009). Areas in a darker shade indicate higher levels of carbon. © Crown copyright / database right 2010. An Ordnance Survey / EDINA supplied service.

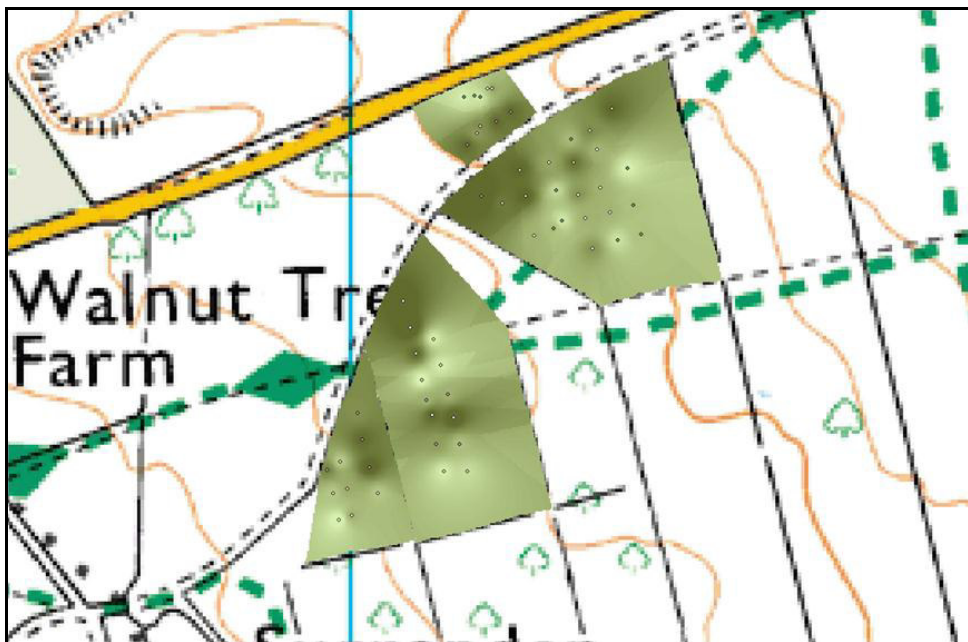


Figure 2.1.8. Site 8 map indicating soil C using data collected (the dots on the map) and interpolated to create a continuous surface using the inverse distance weighted (IDW) method on ArcMap (ArcGIS version 9.3.1 ESRI, 2009). Areas in a darker shade indicate higher levels of carbon. © Crown copyright / database right 2010. An Ordnance Survey / EDINA supplied service.

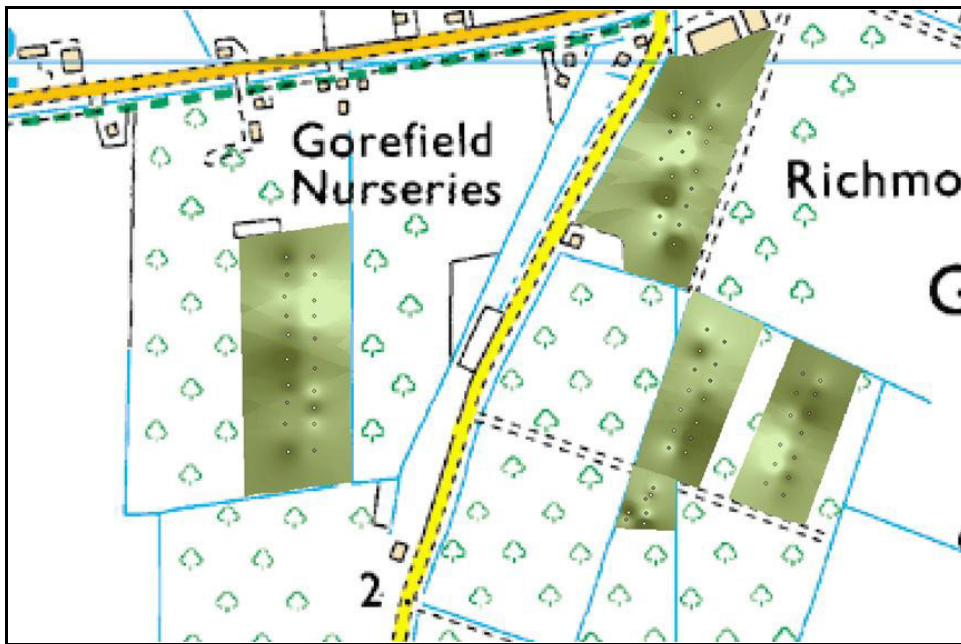


Figure 2.1.9. Site 9 map indicating soil C using data collected (the dots on the map) and interpolated to create a continuous surface using the inverse distance weighted (IDW) method on ArcMap (ArcGIS version 9.3.1 ESRI, 2009). Areas in a darker shade indicate higher levels of carbon. © Crown copyright / database right 2010. An Ordnance Survey / EDINA supplied service.

Appendix 6.1: Apple Orchard Carbon Footprint Questionnaire

BANGOR UNIVERSITY – CARBON FOOTPRINT OF HORTICULTURAL CROPS

The purpose of this questionnaire is to collect data that will enable the carbon footprint calculation of the orchard system for my research into calculating carbon stored in orchards. It is appreciated that detailed information is being requested here, but an accurate and representative carbon footprint must be based on detailed and accurate data.

- Please clearly state if a question is not applicable to this farming system by writing n/a in the space provided.
- If the question is applicable to this farming system but the answer is unknown, please clearly state by writing unknown.

The first half of the survey deals with field operations; please provide all data per hectare of cropped area per year. If for some reason you cannot do this then please make it clear what units you are using, e.g. acres or for the whole farm. The second part of the questionnaire deals with post-harvest operations. Please provide as much detailed information as possible in order to increase the accuracy of the carbon footprint calculation.

If you have any queries or questions when filling in this questionnaire please contact Rosie Anthony by e-mail afp861@bangor.ac.uk or on 01248 383704.

1. Contact Information and Description

1. Farm name

2. Farm address

3. Farm postcode or co-ordinates

4. Brief description of soil type

5. Name of contact person at the farm

6. E-mail address of contact person at the farm

7. Telephone number (including international code) of farm contact

8. Crop name and variety

9. Area currently under this crop (ha)

10. Average annual harvested yield (tonnes of this crop from the whole farm)

11. Average annual marketable yield (tonnes of this crop from the whole farm that is marketable and goes to retail)

12. How long has this land been under agricultural production (years)?

13. If less than 20 years, what was the land used for before 1990? (eg dairy, forest, grassland)

2. Planting

14. Please briefly describe the planting process (the subsequent questions will enable more detail to be give)

15. What is the average lifetime of a plant/tree on this farm?

16. Are seeds/young plants sown? Please specify any other

17. How many seeds/plants/trees are there per hectare?

18. Are young plants: (please write "yes" if this applies and the proportion e.g. 40%)

Propagated on this farm		
Bought in		

19. If bought in, how old are they when bought (and units e.g. months/years)?

20. Where are they bought from (city or postcode)?

21. How are they transported? Is this refrigerated?

22. Are young plants/trees treated for disease before planting out (please tick either yes or no)?

Yes	
No	

23. If yes, please state the product/chemical used, rate applied at and the average amount used per year

Chemical name and formula	Amount and units (e.g. kg, litres) used annually

24. Do they arrive to the farm in growing media?

25. If yes, what type of media is it, and how much is used per plant?

26. What type of containers do they arrive in? Please state material made from and weight of each unit

27. Are containers re-used? If so, how many times?

Yes/No	Number of times re-used

28. What happens to these containers at the end of their life?

29. Other comments not included above

3. Irrigation

30. Please provide a brief description of irrigation used for this crop on this farm (cannon, gravity, spray etc.), including the possible variations within the year and from year to year

31. How much water is typically used in one year on a hectare of this crop?

Irrigation type (e.g. drip, pivot)	Average amount of water used (units)	Max.	Min.	Units

32. How are the irrigation pumps powered? (e.g. electricity, diesel)

33. What is the total energy used for irrigating this crop? Please specify units e.g. kWh/m³

		Units
Electricity		
Diesel		

4. Fertiliser Application

34. Operation (e.g. base dress, top dress, foliar etc)

Details (Fertiliser product name and chemical formula if known)

Amount applied annually (rate and units e.g. kg ha⁻¹)

Machines used

Other (if compost is used, specify % N and components breakdown)

35. Other comments not included above

5. Pesticides (Pest and disease management, regulators and weeding)

36. Please give details of pesticide usage:

Type (e.g. fungicide, herbicide, soil fumigant, growth regulator, defoliant etc)	Details (commercial name, % active ingredient)	Amount applied annually (amount and units e.g. per hectare)	Machines used

37. Other comments not included above

38. Please provide a brief description of the weeding operations undertaken for this crop (e.g. hand-weeding, use of herbicides etc)

39. Please briefly describe any machinery used for weeding

40. Other comments not included above

6. Sundry Inputs

Are other materials used in growing this crop? It is important to understand how much of various inputs such as plastic fleece, plastic bunch covering, plastic trays etc. are used. Some of these items may have many functions on the farm, and may last for many years; please provide information about any items which have a lifetime of 5 years or less. When completing this part of the questionnaire, it may help to think about how many of each item are bought new per year.

In this section, please provide details of any sundry inputs used for production and harvest of this crop. Examples of what may be included:

- Plastic sheeting/mulch
- Pots and trays
- Growing media (e.g. peat, compost, rockwool etc)
- Plastic or cardboard boxes for carrying inputs to/from field

41. Please state any sundry items used

Material made from (e.g. PVC, cardboard, wood, polythene etc)	Number/amount used annually (and units)	Number of years or times used before replacing

42. Please state the fate at end of life for each sundry item listed (e.g. recycle, landfill etc)

Sundry Item	Fate at end of life

43. Other comments not included above

7. Harvest

44. Please describe the harvest operation of this crop

45. Please state the distance from field to grading area and from grading area to dispatch (if other locations are involved in the transport of crop from field to dispatch, please include distances here) and the vehicle used for this transport (and % of product travelling this way)

46. Are any sundry materials used during harvesting? If so, please detail

47. Are any ripeners or ripeness suppressors used?

48. Is machinery used for harvesting?

49. Other comments not included above

8. Residues

50. What is the estimated annual quantity of crop residues (such as prunings, fallen leaves and apples) and units (e.g. kg/ha)?

51. Please state the fate of these residues and the proportion which see this fate (e.g. composted, burned, ploughed into soil, dumped, left on the field etc)

Example: Prunings – left on the ground 100%

52. Other comments not included above

9. Machinery

Please describe the fuel use of ALL machines used in the production of this crop.

53. Please list total amount of fuel used for this crop annually or provide a breakdown of each machine (owned by the farm and leased) for each operation:

- Soil preparation
- Fertiliser and pesticide application

- Ripeners or ripeness suppressor application
- Installing/removing irrigation equipment
- Harvesting operations
- Grubbing operations

State machine name/description, fuel type, operation type and consumption (with units)

54. Other comments not included above

10. Sort, clean, cool

This part of the survey deals with post-harvest operations; when answering questions, please state clearly the units of measurement, e.g. per tonne, per hour etc. Post-harvest operations are a vital part of the carbon footprint of food products. Please therefore provide as much information as possible

55. Does the harvested product undergo a pre-storage sorting/cleaning/grading process post-harvest? If no, go to Q59 next.

56. If yes, describe the process

57. Volume of crop cleaned/selected annually

Brief description of machine used

Energy source (e.g. electricity, diesel)

Annual energy use (value and units)

58. Is field heat actively removed from newly harvested crops?

If no, go straight to the next section "Extra Information"

Yes/No

59. If yes, please provide details of the method used and relevant machinery.

Please state whether powered by diesel, electricity etc. and give annual energy use values and units.

60. Any other information not included above

11. Extra Information

61. If you would like to provide any further information about this product's supply chain which may be of help to the carbon footprint, please use the box below to do so.

THANK YOU FOR COMPLETING THIS CARBON FOOTPRINT QUESTIONNAIRE.

Appendix 6.2: Orchard Fertiliser Details

Table 6.2.1. Key for Table 6.2.

Number	Fertiliser (% active ingredient)
1	Aminosol (9% organic bonded N)
2	Ammonium nitrate (35% N)
3	Bittersalz Epsotop
4	Bittersalz Magnesium Sulphate (16% Mg, 13% sulphate)
5	Bortrac (10.9% boron, 4.7% N)
6	Bud builder (24% Mg, 10% Zn, 6.9% P ₂ O ₅ , 3% B)
7	Calciphite (15% P ₂ O ₅ , 5% K ₂ O, 8% Ca)
8	Calcium chloride
9	Calcium flake
10	CAN
11	CN (calcium, nitrate)
12	Carnival (15% Ca, 2% Mg, 10% N)
13	Epsotop (15% MgO, 31% SO ₃ , 0.9% B, 1% Mn)
14	Headland Boron (15% boron ethanolamine)
15	InCa
16	Keserite
17	Krista-Map
18	Lime
19	Magnesium sulphate
20	Mag Super 80
21	Manganese 500 Headland
22	Manifol advance
23	Mantrac (50% manganese)
24	MAP
25	Maxicrop triple
26	Muriate of potash (60% K ₂ O)
27	Nitram (34.5% N)
28	Nutriphite (10% calcium compounds)
29	Opte-phos
30	Potassium bicarbonate
31	Potassium nitrate
32	Seaweed
33	Seniphos (23.6% P ₂ O ₅ phosphorus, 3% calcium)
34	Solubor (17.5% boron)
35	Stopit (12.1% calcium chloride liquor)
36	Sulphur F3000 (34% S, 14.8% N)
37	TSP triple super phosphate (48% P ₂ O ₅)
38	Urea (46% N)
39	Wuxal Top P
40	Zintrac 700 (70% Zn)
41	20:10:10 (N:phosphate:potash)
42	33.5% N

Table 6.2.2. Fertiliser type and amount used for each orchard site.

Fertiliser (kg ha ⁻¹ yr ⁻¹)	Orchard															
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
1										2.46	2.46	0.62				2.46
2	75.00								45.00	145.00	137.5	50.00		150.00	300.00	250.00
3			20.00	20.00	20.00		20.00	10.00								
4									6.25	6.25	12.5	12.50				
5																4.09
6														5.00	10.00	
7														8.00		
8		50.00	13.7	13.70						23.25	24.95	5.00	60.00			
9													8.00			
10													150.00			
11									12.5	12.5	12.5	12.5				150.00
12			34.5	34.50	34.50	4.5	19.5	34.5								
13							5.00	5.00							26.25	
14			2.69	2.69	2.69	2.68	2.68	2.68								
15			3.00	3.00	3.00											
16			100.00													
17									9.00	9.00	6.00	12.25				
18					3297.0 0	3468.0 0		5430.00			1000.0 0					
19		225.00														
20			3.99	3.99	3.99	11.97	3.99	11.97								
21										0.3	0.30					
22									0.25	1.1	1.60	0.5				0.9
23																1.83
24														25.00	50.00	9.00
25			2.00	2.00										15.58	6.68	
26			100.00				133.33			45.00						

27		360.33	360.33	360.33		174.00	87.00								
28													1.00		
29										10.00					
30											3.00				
31												10.00			14.19
32															2.04
33	65.6													19.68	
34								0.50	0.50	0.50					
35	119.43	39.81	39.81	39.81		13.27	26.54								
36															5.40
37								200.00	100.00	100.00	100.00				
38		30.00	20.00	20.00	20.00	20.00	20.00	68.75	68.75	72.50	62.50	28.00	11.00	49.00	26.00
39			7.80	7.80	7.80	20.80	11.7	7.8							
40			1.734	1.734	1.734	1.734	1.734	1.734		0.61	0.78	0.43			1.734
41		125.00													
42															300.00

Appendix 6.3: Orchard Pesticide details

Table 6.3.1 Key for Table 6.3. Pesticides used.

Number	Pesticide (% active ingredient)
1	Agrovista Paclo (25% paclobutrazol)
2	Agrovista Reggae (48% thiacloprid)
3	Agrovista Trotter (24% methoxyfenozide)
4	Alpha Captan 80 WDG (80% captan)
5	Aphox (50% pirimicarb)
6	Bellis (25.2% boscalid, 12.8% pyraclostrobin)
7	Calypso (48% thiacloprid)
8	Captan (48.9% captan)
9	Codacide
10	Companion Gold
11	Coragen (20% chlorantraniliprole)
12	Cuprokylt (50% copper oxichloride)
13	CYREN (48% chlorpyrifos)
14	Dithianon WG (70% dithianon)
15	Dithianon flowable (75% dithianon)
16	Dodifun 400 SC (40% dodine)
17	Equity (48% chlorpyrifos)
18	GIBB Plus (1% gibberellins)
19	Headland Inorganic Liquid Copper (43.5% copper oxychloride)
20	Indar (5% fenbuconazole)
21	Insegar (25% fenoxycarb)
22	Kindred (35% meptyldinocap)
23	Maccani (12% dithianon, 4% pyraclostrobin)
24	Mainman (50% flonicamid)
25	MaxCel (1.9% 6 benzylaminopurine)
26	Mix mate
27	Nimrod (25% bupirimate)
28	Novagibb (1% gibberellins)
29	Optica
30	Pitstop
31	PP Captan 80 WG (80% captan)
32	Radspor FL (45% dodine)
33	Regalis (10% prohexadione calcium)
34	Regulex 10 SG (10% gibberellins)
35	Robut 20 (20% myclobutanil)
36	Runner (24% methoxyfenozide)
37	Scala (40% pyrimethanil)
38	Spray Guard
39	Spryte Aqua
40	Steward (30% indoxacarb)
41	Stoby WG (50% kresoxim-methyl)
42	Switch (37.5% cyprodinil, 25% fludioxonil)
43	Systhane 20 EW (20% myclobutanil)
44	Topas (10% penconazole)
45	Topenco 100 EC (10% penconazole)
46	Transcend
47	X-Change

Table 6.3.2. Pesticide type and amount used for each orchard site.

Pesticide (kg ha ⁻¹ yr ⁻¹)	Orchard															
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
1									0.50	0.50	0.50	0.25				
2									0.25	0.25	0.25	0.25				
3									0.25	0.25	0.63	0.25				
4			1.00	1.00	1.00	9.00	12.00	10.00	4.00	2.00	2.00	2.00				
5													0.22			
6	3.20	0.80	1.60	1.60	1.60	0.80	1.60	0.80			1.60	0.80		0.80	0.80	0.80
7		0.30					0.36						0.81	0.45	0.45	0.60
8	9.00	12.60											5.00	6.00	9.50	
9														4.63	9.25	
10										0.35	0.30					
11	0.38		0.38	0.38	0.35	0.38	0.38	0.38	0.21	0.21	0.38	0.38	1.19	0.20	0.20	0.38
12	1.00	1.00							1.25	1.25	1.25	1.25				1.00
13							1.18		1.08	1.08	1.08	1.08		2.17	4.34	
14	4.74	4.74	2.25	2.25	2.25	0.75	0.75	0.75	1.50	1.50	2.25	2.25	2.00			
15			6.32	6.32	6.32	1.74	1.74	1.74					1.58	1.58	4.74	
16											2.00	2.00				5.00
17	3.00	5.00	2.14	2.14	2.14	2.14	0.97	2.14								
18									0.52	0.52	0.77	0.77				
19							3.38	3.38								
20													0.63			
21													0.40			
22														1.16	2.33	
23							2.50				2.00	2.00				
24			0.28	0.28	0.28	0.14	0.14	0.14						0.14	0.28	0.00
25			7.50	7.50	7.50											
26	1.00	1.00											2.13			
27		0.64												0.06	0.11	
28		0.26												0.94	0.94	
29															2.00	
30													4.00			
31									16.5	20.00	15.5	12.5				12.00
32			1.00	1.00	1.00	1.00	2.00	1.00						4.00	4.00	
33	2.50	2.25	2.40	2.40	1.40			1.20					1.06			
34		0.60				0.03	0.04	0.03								
35									1.65	1.65	2.03	2.03	0.40			1.20
36		0.53	0.85	0.85	0.85	0.85	0.85	0.85						0.64	0.64	
37	1.34	1.34	1.07	1.07	1.07	0.54	1.07	0.54	4.28	4.28	3.21	3.21		0.54	1.61	2.14

38									0.31	0.10	0.31	0.31	0.16				
39			2.00	2.00	2.00	2.00	2.00	2.00									1.50
40	0.20	0.20	0.25	0.25			0.25	0.25					0.20				
41						0.40	0.20	4.00						0.40	0.40		
42										0.80							
43		0.68	2.04	2.04	2.04	2.04	3.18	3.06					0.41	1.39	1.39	0.52	
44	1.60	1.70	1.50	1.50	1.50								0.80	1.50	2.00	0.50	
45																0.85	
46									2.51	2.51	2.51	2.20				0.94	
47			3.25	3.25	4.50			1.00									

Appendix 6.4: Orchard Herbicide Details

Table 6.4.1. Key for Table 6.4 list of herbicides used.

Number	Herbicide (% active ingredient)
1	Cleancrop Gallifrey 200 (20% fluroxypyr)
2	Clinic Ace (36% glyphosate)
3	Cultar (25% paclobutrazol)
4	Depitox (50% 2,4-D)
5	Glyphogan (36% glyphosate)
6	Glyphosate (36% glyphosate)
7	Harvest (15% glufosinate-ammonium)
8	Headland Trinity (26.8% MCPA, 4.25% mecoprop-p, 1.53% dicamba)
9	Hormone
10	Nufosate Ace (36% glyphosate)
11	Roundup biactive (36% glyphosate)
12	Roundup (36% glyphosate)
13	Starane 2 (20.7% fluroxypyr)
14	UPL Campnex (7% mecoprop-p, 5.6% dichlorprop-p, 4.5% MCPA, 2.9% 2,4-D)
15	Weedazol-TL (22.5% amitrole)

Table 6.4.2. Herbicide type and amount used for each orchard site.

Herbicide used (L ha ⁻¹)	Orchard															
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
1									0.50		0.50	0.50				
2										2.50	2.50					
3	1.70	0.45						0.60	0.25	0.25	0.25			0.60	0.45	
4															2.00	2.80
5																3.00
6													6.00	5.00	10.00	
7																3.00
8			4.50	4.50	4.50		4.50	4.50								
9													6.00			
10			4.00	4.00	4.00	4.00	4.00	4.00								
11			4.00	4.00	4.00	4.00	4.00	4.00								
12	6.00	6.00														
13			0.80	0.80	0.80	0.80	0.80	0.80								
14									1.50		1.50	1.50				
15									3.50		3.50	3.50				