Terrestrial carbon storage and sequestration potential of institutionally managed estates

By

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Abstract

Many institutions have declared a climate emergency and are committed to ambitious net-zero carbon aims. However, few institutional carbon management plans consider the terrestrial carbon store of the estate in a quantitative or qualitative way. Using Newcastle University as a case study, this research demonstrated ways to quantify and potentially augment the soil and tree carbon stocks of institutional estates by changes in land management. The terrestrial carbon store of Newcastle University's estate was quantified with field work, and scenarios of the off-setting of institutional carbon emissions were derived by considering alternative land management. Additionally, the application of wheat straw biomass and its biochar to urban soil was investigated for carbon sequestration. To quantify the current carbon storage baseline of the institutional estate, soil and tree carbon was surveyed for two research farms, campus green spaces, and a sports field. The total terrestrial carbon stock of Newcastle University's estate was found to be 17 times greater than the annual institutional CO_2 equivalents-C emissions in 2019-20. Newcastle University could off-set half of its institutional CO_2 equivalents-C emissions over a period of 40 years by converting its farms into woodland. Reverting farm management to practices shown on old maps from 1900 with more permanent grasslands could off-set 64 percent of institutional CO_2 equivalents-C emissions over a period of 5 years. Other measure such as doubling the number of free-standing trees on the farms, converting all lawns on the central campus into urban woodland, or amending the Newcastle Helix brownfield reclamation site soil with 2% biochar would off-set less than 3 percent of institutional emissions over 40 years. Interviews with estate, farm and carbon managers revealed reluctance to accept the dramatic land management changes which would be needed for tangible off-setting of institutional emissions, but it will be difficult to achieve net-zero carbon emission aims without serious consideration of off-setting opportunities in Newcastle University's estate.

Declaration

This thesis has been written by Jiaqian Wang, and has not been submitted previously for applying any other degrees. All sources of information have been acknowledged explicitly by reference.

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1. Chapter One. Introduction

1.1 Context

The world has already been warmed by 1℃, with a range of 0.8-1.2 ℃, largely due to the burning of fossil fuels. The Fifth Intergovernmental Panel on Climate Change Report has appealed and urged the individual members to make efforts to alleviate the exaggeration of global warming which should halt a rise of 2℃, but are attempting to reach only a 1.5℃ increase against the pre-industrial period. The detrimental impact of abnormal global warming has already occurred, covering water stress, degraded habitats, the loss of biodiversity, extreme weather events, etc, and further limits crop productivity and result in food shortages as well as an unequal social-economic development between different countries (Masson-Delmotte et al., 2018, Shukla et al., 2019). Additionally, heat waves in cities affect the longevity of fundamental urban infrastructure, and pose risks to human well-being, for example malaria and dengue fever will quickly spread under high temperatures (Masson-Delmotte et al., 2018). The greenhouse effect is closely related to global warming. The normal functioning of the greenhouse effect is essential and paramount to maintain stable living habitats for creatures. The considerable amount of greenhouse gases emitted by anthropogenic activities, however, hampers the effect of maintaining the heat of the natural ecosystems. Amid all contributors resulting in anthropogenic greenhouse gases which result in global warming, carbon dioxide concentration in the atmosphere accounts for the majority where the contribution from CO_2 versus Non-CO_2 is 71.6% and 28.4%, respectively (Bernstein et al., 2008). The natural dynamics of carbon transportation and reactions are nowadays disturbed by considerable anthropogenic energy consumption, the rate and scale of urbanisation, the changing land use, and the constant development of industries and human activities. The terrestrial carbon sink plays an important role in the Earth's carbon cycle, where the geological pool is the second largest carbon pool among a total five C pools in the Earth's system (Lal, 2003). One critical option to moderate the current increase in atmospheric CO_2 is to maximize the carbon sequestration of different land uses and to augment the terrestrial carbon stock by land transformation.

Numerous institutions have proposed net-zero carbon strategies associated with a carbon management plan to contribute towards the 1.5 ℃ target. An investigation, which examines the carbon neutrality schemes from 327 different cities across twenty-eight European countries, has revealed that 78% of the participating sectors plan to achieve the carbon reduction target with a climate change plan (Salvia et al., 2021). The United Kingdom (UK) government is committed to achieve net-zero of greenhouse gases by 2050 and will host the

26th United Nations Climate Change Conference in Glasgow (COP 26) in November 2021, which is an opportunity to present real action in climate change mitigation among the indigenous areas (Stark et al., 2019). At the same time, post Brexit, the UK is likely to divest itself of constraints of the previous European principles, hence some residents are calling for a more noticeable emissions reduction. In the past few years, various departments of the UK government have been engaged in scaling up new policy developments, initiating advanced carbon removal technology as well as reforming the supply chains with low carbon production.

To achieve the goal of carbon neutrality, national governments and local councils need support, investment, cooperation and experience from institutional bodies which can launch short term initiatives in the individual industries while aligning with long-term goals. For instance, the Confederation of Passenger Transport has drawn up a plan that all new buses are expected to have ultra-low, or zero emissions by 2025 (Climate Change Committee, 2020); the Office for Low Emission Vehicles has offered funding to 28 councils across the UK to build more public charging points in the proximity of residential areas which encourages the purchase of electric vehicles (Stark et al., 2019). The 25 Year Environment Plan looks forward to optimising the delivery of "Green Brexit" and is evaluating the restoration efficacy of woodlands where it specifically aims to conduct the sustainable management over England's soils by 2030 as well as extend a Nature Recovery Network accompanying the enhanced protection of existing trees (Defra, 2018).

Around 2010, the higher education communities have agreed that universities should take action on easing climate change, because the carbon emissions resulting from the university academic activities are substantial. Students would be involved in many scientific projects, which are significant pathways to test the practicality and acceptance of new carbon removal technologies that are designed to make a profit in the markets (Lewis and Patton, 2010, Robinson et al., 2018). Furthermore, the change, led by universities, can become a role model for wider society. At present, the higher education sectors across the UK exceed 2.31 million people, with almost 0.34 million staff and 1.97 million students, covering 161 institutions (HESA, 2019). Similarly, the higher education organisations are one of the largest landholders, occupying 12,456 hectares of land, which contains 6,926 ha of ground area and 1,482 ha of playing fields area (HESA, 2019). As a result, at the universities a large number of education opportunities can assist practitioners to test carbon management standards and influence student behaviour (Robinson et al., 2018). Though 65% (totalling 126) of universities had struggled to reach the 43% carbon reduction target by 2020 against the baseline in 2005/06, which is the requirement of Higher Education Funding Council for England, there are still 87 colleges or universities which accomplished a decrease of carbon emissions ranging from 0.34% to 46.54% over the 8 years since 2005. Plus, 44 institutions believed they could achieve the 43% reduction target in 2020 (Brite Green, 2014). Since the current carbon management reports from universities mainly quantify the carbon emitted by academic activities and the operation of basic systems in buildings, the knowledge about how much carbon is already sequestered in the terrestrial land owned by universities is limited. And the different performance of carbon storage as land use changes, and the potential of extra carbon sequestration due to land conversion with the changing space usages, have not yet been demonstrated within a university scale. In this context, Newcastle University aims to achieve net-zero carbon emissions by 2030. Its main campus is located in the core of Newcastle City and it has two farms are owned by the university, which makes Newcastle University the 4th largest farming university in the UK (Farm Diversity). This extent of university-owned land is important in terms of potential carbon accumulation in soil and plants, and carbon sequestration opportunities through better land management encompassing different farming treatments such as ploughing frequency and crop vegetation types.

In addition to land-use change, geoengineering is increasingly considered to address the climate emergency. On the path towards engineered carbon removal and sequestration, biochar is widely used for amending the ground and increasing its carbon content (Dong et al., 2019, Kuppusamy et al., 2016). Biochar is the production of carbonised biomass from pyrolysis with limited O_2 or in the absence of oxygen. The biomass feedstock includes many types, such as crop wastes, animal manures, woody biomass and urban green waste, etc. During pyrolysis, the charring leads to the loss of volatile matter and cracks form in the solid residue due to the shrinkage stresses where C crystals of biomass would be rearranged. Approximately 50% of biomass is left after the charring. The structural or molecular changes during the pyrolysis partly impede the further decomposition of char in an external environment. So biochar stays in the soils for centuries leading to a materially greater amount of C sequestered compared to unprocessed organic matter (Lehmann et al., 2006). Secondly, the porosity of the biochar has substantially increased, resulting in an increase of surface area. Also, the high ash content present in the biochar is responsible for high pH, and thereby influencing cation exchange conductivity. As a result, mixing biochar with soils can bring a preferred habitat for soil organisms. This is because biochar can be a nutrient source and provide living spaces for bacteria in highly porous sites, strengthening soil water holding capacity and soil aeration. Also, its high pH helps to buffer soil acidity, which would be favourable for soil microbial activities in acidic soils (Nele, 2013). All the exhibited properties from biochar impart it an inherent environmental value as compared to materials that may otherwise be disposed of as “wastes” without discovering the potential benefits of amending or improving soils. Carbon cycling predominantly occurs between the atmosphere and plants, while the terrestrial carbon pool is relatively stable. Based on this context, there is a prediction that, annually, using 1% of carbon intake from plants to produce biochar which is then applied to soils would achieve a 10% mitigation of anthropogenic carbon emissions (Joseph and Lehmann, 2009). In the UK, carbon abatement has an overarching economic value in biochar deployment (Shackley et al., 2011). More specifically, the maximum carbon storage rate resulting from biochar and carbonate materials is 7 Megatonne (Mt) per year within the UK (Renforth et al., 2011). If half of national biochar productivity is used for carbon sequestration in agricultural land across the UK, preferably biochar derived from non-virgin waste biomass of organic feedstocks without chemical or biological processing, an annual 1-6 Mt C abatement in the UK context would be accomplished (Shackley et al., 2010). In the USA, a study conducted over 6 years has stated the increased soil organic carbon is twice as high in mixed wood biochar-treated soils than non-biochar soils (Blanco-Canqui et al., 2020). Placing biochar in actual fields to track the carbon balance change over time will be key to demonstrate the feasibility of biochar technology to achieve carbon sequestration under a real environmental condition with the effects of rainfall and wind weathering.

This thesis investigates a number of critical parameters to assess the current carbon storage of the urban and rural estate managed by Newcastle University, related carbon sequestration opportunities by changing land use, and the feasibility of biochar deployment to combat the climate crisis based on a lysimeter experiment. The work contained in this thesis contributes knowledge across soil and agriculture science, forestry, and biochar technology. It has implications for the interpretation of other studies looking to understand carbon stock in terrestrial lands and waste recycling use within institutional management.

1.2 Scope, Aims and Objectives

This study focuses on carbon abatement activities that are particularly relevant to an institutional such as Newcastle University, and the outcome of field monitored data and associated analysis that could combined with the climate action plan of Newcastle University to frame or support future amendments. In addition to field and laboratory work, information was gathered through the interview of key stakeholders to shed light on the land management challenges and difficulties related to carbon removal approaches. As part of the consultation, we have suggested several options regarding to land conversion for farm directors and estate managers and ask how well they seem to be able to adopt these, what considerations appear to be the most critical to accept this advice and proposed some tentative ways that their comments be addressed.

The major aim of this research is to determine the carbon stock from the land owned by Newcastle University with its potential given the possible changes to land use, and at the same time, to evaluate whether wheat straw biomass or its biochar can sequester carbon in urban soils. The study will therefore provide a baseline for comprehending the factors influencing these carbon variations in the organic waste-amended environment.

The above aims are achieved from a range of field works and measurements through the following specific objectives:

1) Soil carbon classification and vegetation carbon characterization across farm and campus sites (Chapter 3&4)

- The first survey included the determination of organic, inorganic and total C for soil samples from agricultural land and woodland at Nafferton Farm and Cockle Park Farm, and topsoil from the central campus of Newcastle University, with respect to the various land uses associated with historical records about cropping or planting.
- Determine the tree carbon content of plants in the woodlands and hedgerows in farms, and in the urban campus. Woodlands can be classified by leaf types and the planting date, which facilitates the comparison of tree carbon from different groups. On campus, the trees were categorized using diameter at breast height (DBH), height, mature age and carbon storage performance.

2) Determine the function of carbon sequestration of biochar application using landscaping soils (Chapter 5)

- Two lysimeters were established with urban construction sites soils, and amended by wheat straw pellets and wheat straw biochar, respectively. Carbon changes in the two different environments and the related contributors were analysed.

3) Quantify efficacy and capacity of carbon capture by altering land use and the linked management considerations (Chapter 6)

- Alternative land conversions were designed and subjected to analysis to assess its feasibility and the estimated carbon storage pay-back in terms of period and amount.
- Feedbacks from interviews with directors from Newcastle University responsible for land management were reviewed.

1.3 Thesis Overview

Chapter 2 considers the overall knowledge about greenhouse gas effects and summarises evidence that terrestrial land is a fundamental and vital carbon pool. The alteration of land use and its changes would have diverse effects on the function of the ecosystem as well as carbon sequestration. Among all carbon removal approaches, land conversion is the nature-based solution, and the various opportunities for carbon capture with land changes will be listed. Alternatively, biochar application which can be an option to reuse wastes as well as improve soils to store more carbon, the mechanisms of biochar on conditioning the soil environment and how outcomes would differ for various approaches, have been discussed. The chapter ends by evaluating the constraints of policy and assessments on the carbon removal market.

Chapter 3 considers the soil carbon from agricultural land and woodlands as well as tree biomass carbon at two university-managed farms (Cockle Park Farm and Nafferton Farm). Combined with the records relating to crop rotation, tillage practice and woodland development, the results of this chapter show the capacity of carbon sequestration from various agricultural ecosystems and how to predict the potential of carbon sequestration under four scenarios for future land management change at the farms where each aims to reach one specific carbon reduction target.

As a complement to Chapter 3, Chapter 4 investigates the soil carbon on the central campus of Newcastle University, including lawns with some free-standing trees, the woodland in the urban area and a university-managed sports field in a suburban area. Meanwhile, total biomass carbon storage from all trees on the campus of the university has been calculated and categorized with species, physical dimensions, and tree ages. Chapter 4 considers the variation of tree carbon as the per unit tree cover or as the single plant, and how tree cover influences soil carbon. The terrestrial carbon storage ability per canopy and land cover in the campus of Newcastle University can be representative of Newcastle city, and scenarios were developed to predict how much carbon can be sequestered over the entire urban green ecosystem.

Chapter 5 presents a study on monitoring the carbon store in two lysimeters for three years (2018-2021). Top soil in one of them had wheat straw biochar applied at the start, and for comparison the other was amended by wheat straw pellets. Both were set up in the campus of Newcastle University and used to investigate man-made landscaping soils typically found in urban redevelopments. Over the experimental period, soil carbon, carbon dioxide gas emissions from three different soil depths, leachate carbon and vegetation carbon have been measured and calculated to compare the difference of carbon storage in soil between the two treatments.

Chapter 6 concludes the compiled results from Chapters 3, 4 and 5 because all of them are complementary in demonstrating and comparing the magnitude of carbon storage from the farm fields and the city area owned by the university in northern-eastern England, and feasibility of biochar application to mitigate carbon emissions across the campus. The remaining research questions and highlighted improved points are expressed in this final chapter.

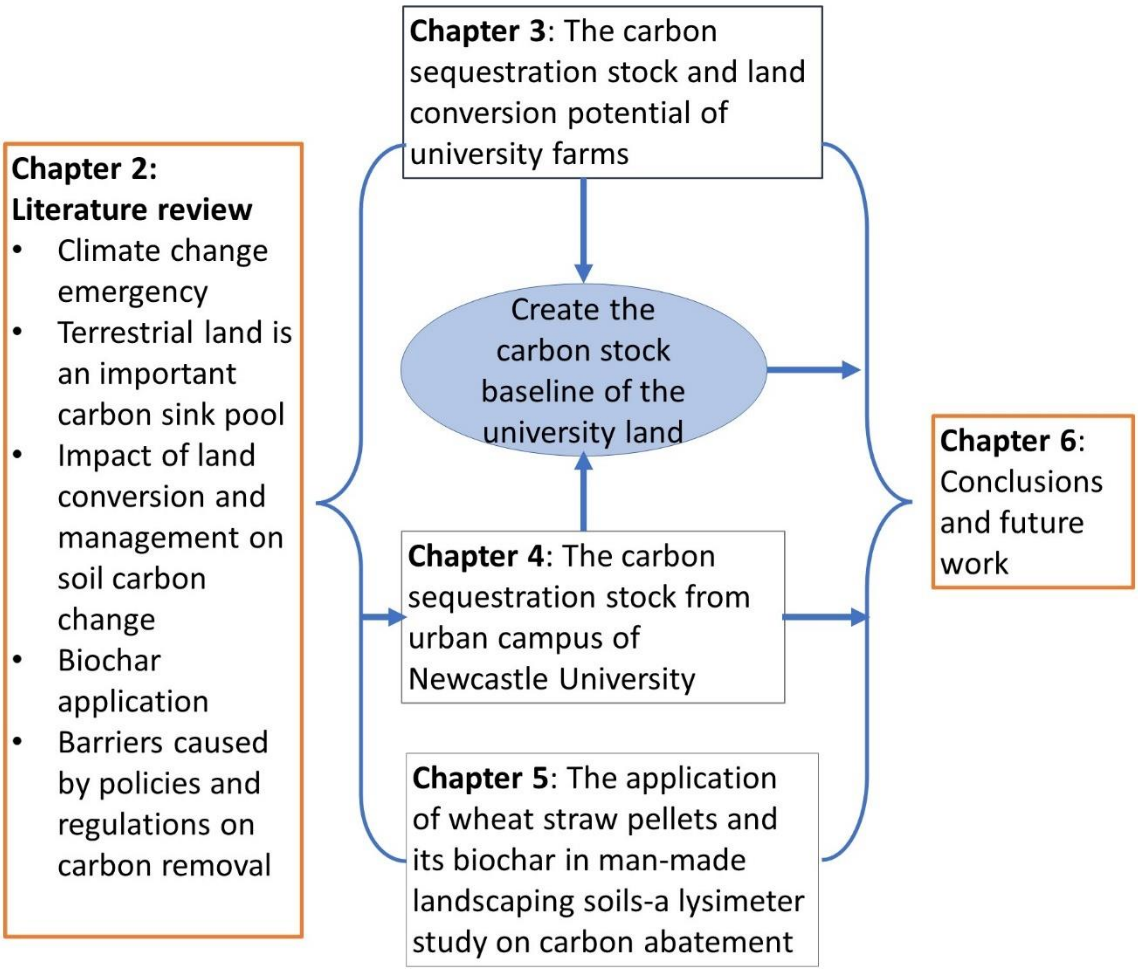

Figure 1. 1 Thesis Overview

2. Chapter Two. Literature Review

2.1 Chapter overview

The first part of this chapter considers the origin of climate change and its relationship to Greenhouse Gas Emissions, particularly carbon emissions. Carbon is one of the most dominant elements in the biosphere and its ecosystem and how its compositions and substances occur should be considered in order to study the carbon cycle. In the carbon cycle, terrestrial land, including vegetation, plays a principal role in storing or sinking C where its capacity as an annual carbon sink is 33% higher than that of the ocean from 2008-2017 (Le Quéré et al., 2018), and the related carbon transportation process is described here. Then, the impact of different types of landscape on carbon sequestration and the related management policies are discussed. Also, various approaches for combating carbon emissions are summarized, of which biochar application will be introduced in section 2.6. Moreover, the connection and cooperation among multi-level departments or institutions in national regions are compared. Finally, all key points from literature, being in accordance with the necessity to set out our whole project framework, are highlighted.

2.2 Climate Change and Greenhouse Gases Emissions

2.2.1 A brief history of climate change and global warming

The greenhouse effect is an essential feature of the Earth to maintain a suitable environment and thereby supporting the survival of living things. The greenhouse effect is one of the fundamental processes to adjust the climate by the alteration of longwave radiation from the Earth's surface back towards the atmosphere due to the presence of greenhouse gases (GHG) (Le Treut, 2007). Figure 2. 1 illustrates the circulating pathways of GHG among various ecospheres. Based on this figure, it can be observed that the circulation and interaction among various greenhouse constituents play a primary and dominating role on the energy exchange in every environment or the survival of every biological species.

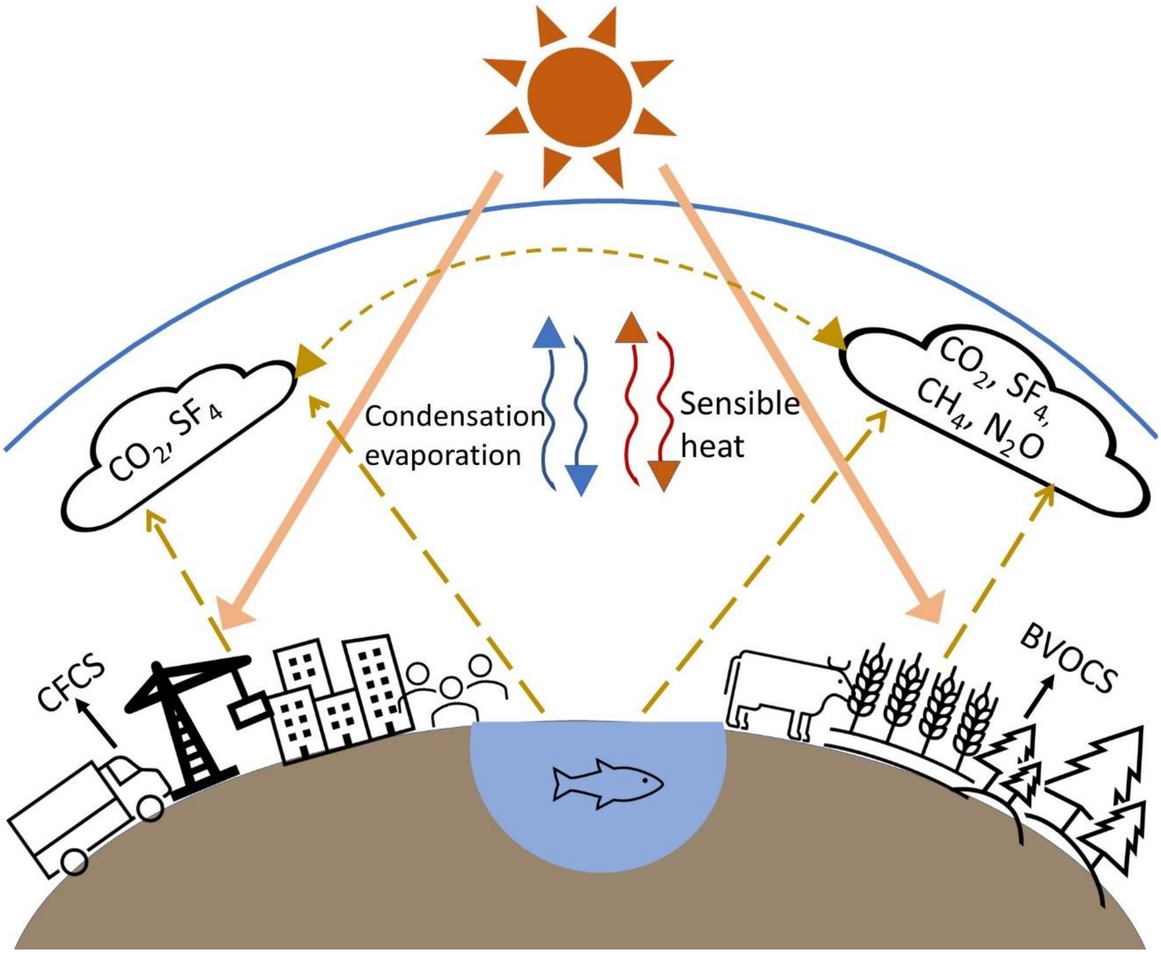

Figure 2. 1 The diagram illustrating the cycling of greenhouse gases. CFCS: Chlorofluorocarbons; BVOCS: Biogenic volatile organic compounds. (Produced by Jiaqian Wang).

The earliest analogy to the greenhouse effect can be traced back to the 18th century. Horace Benedict de Saussure set up a “heliothermometer” to simulate the raising temperature in a darkened box covered by panes of glass (Le Treut, 2007). In 1856, Eunice Foote’s experiment suggested a strong relationship between atmospheric gases and climate change (Jackson, 2020). Three years later, Tyndall (1861), not aware of Eunice Foote’s work, also pointed out that atmospheric constituents such as water vapour and CO_2 would result in the greenhouse effect where water vapour is the predominant index, e.g. the humid low-latitude areas experience a larger greenhouse effect because of the enriched water content in the air (Le Treut, 2007). In 1896, Svante Arrhenius discussed a raising atmosphere temperature was resulted by CO_2 emissions from burning fossil fuels (Weart, 2008). Later, by observing a number of weather stations globally and processing a set of formulae, Callendar (1938) demonstrated that the doubling of artificial CO_2 concentrations in the atmosphere can increase the mean temperature nearly 2℃. Furthermore, Charles David Keeling (1961), who conducted a systematic measurement of atmospheric CO_2 in Mauna Loa on the island of

Hawaii, confirmed that monthly CO_2 concentrations were rising from 1957 to 1960. This finding was the first unambiguous proof related to the dynamics of atmospheric CO_2 content in climate science over the world. These landmarks mentioned above, combined with the evolution of recognizing the importance of global warming and the increase of CO_2 concentration, are displayed in Figure 2. 2.

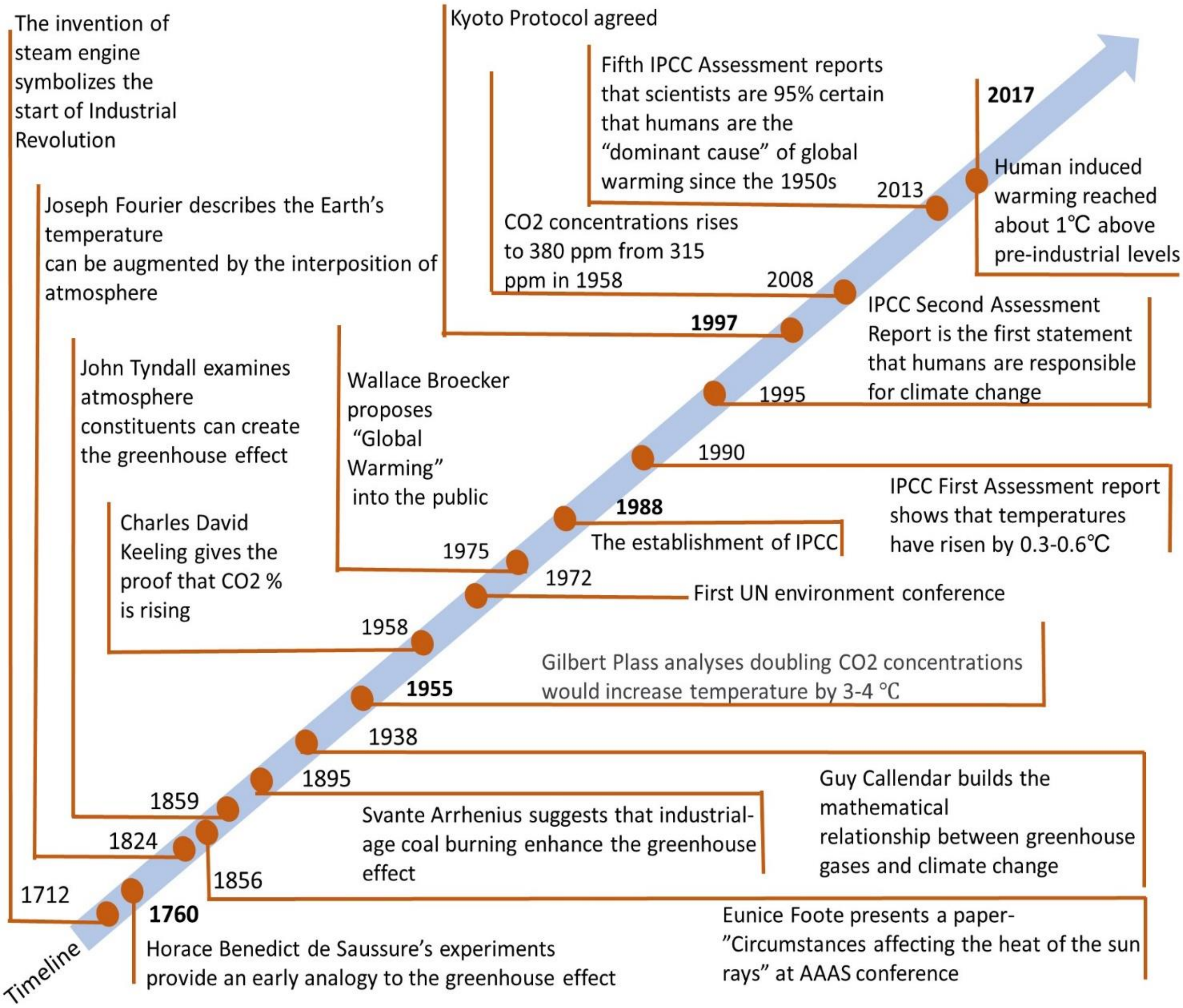

Figure 2. 2 The evolution of study in recognizing the importance of climate change and global warming. (Produced by Jiaqian Wang)

In 1975, Wallace Broecker (Black, 2013) proposed the term "global warming" in a scientific paper. Since then, this terminology has been discussed and researched as a public and international topic, and the past and future trends of global warming have been simulated. Jia et al. (2019) have investigated that the increasing variation of mean land surface air temperature (Δ1.53℃) is larger than change in the mean global surface temperature (Δ0.87℃) from the preindustrial period (1895-1900) to the present time (1999-2018). If this speed continues without adequate policies and regulations that intend to alleviate the warming environment, human-induced global warming will incur a 1.5℃ increase in warming around 2040, which will definitely bring an irreversible threat to all creatures on the planet (Masson-Delmotte et al., 2018). According to the Fifth IPCC Assessment report (2014), preventing the increase of mean global temperature by 2℃ and endeavouring to limit the temperature increase to 1.5℃ against pre-industrial levels are new aims for humankind.

2.2.2 Global threats caused by climate change

Since the 1970s, other greenhouse gases (GHG) apart from CO_2, were recognised: CH_4, N_2O, CFCs (chlorofluorocarbons), atmospheric aerosols, and all these gaseous emissions are primarily caused by human activities; mainly the burning of fossil fuels and deforestation (Le Treut, 2007). Scientists believe that humans are responsible for the massive changes of climate since the 1950s (Black, 2013), though natural climate variability would disturb the local environment change as well, e.g. wildfires, volcanic explosion (Jia et al., 2019). Climate change caused by the greenhouse effects, predominantly referred to as global warming, is a serious issue because a wide range of detrimental impacts have affected the stability of ecosystems, habitats, and atmosphere. For example, the heating trend causes glacial ablation in the Earth's polar regions and high mountains. On the one hand, frozen glaciers are significant natural water reservoirs so the decrease of freshwater storage would threaten the city's water supply. On the other hand, the coastal countries mainly composed by small islands like Maldives, Palau, Comoros, and Tuvalu, will suffer the loss of terrestrial lands, while countries, like Bangladesh, where people are heavily reliant on farming with inadequate infrastructure, and at the same time are located at the low elevation, face the displacement of people due to the rising sea level, frequent cyclones, and the severe land salinisation (Jha, 2021). Besides that, the number of extreme weather events, including hurricanes, frequent storms, droughts, and floods (e.g. EI Nino events), would lead to an increased ecological disturbance (Masson-Delmotte et al., 2018). Concurrently, the global economy, technologies and social-cultural issues are importantly related to climate change. Due to an unequal environmental degradation and deterioration across different regions, the gap of economic growth between the countries with a more stable environment and those that are poor or have vulnerable natural systems is becoming wider (Masson-Delmotte et al., 2018). The rising temperature also can exacerbate infectious diseases, food shortages and biodiversity loss. These astonishingly diverse climate-related events threaten the harmonious development of humans and wildlife. Understanding the core factors and pathways affecting climate change are vitally important before taking steps to curb the deteriorating climate environment.

2.2.3 Development of carbon dioxide technology

Joseph Black who discovered carbon dioxide (CO_2) in the 1750s was a Scot, who published the result when he studied at Edinburgh University (LENNTECH). In 1900, Knut Angstrom, a Swedish scientist, showed that the absorption of CO_2 in the infrared spectrum is very high even for low CO_2 concentrations (Black, 2013). Therefore, if more CO_2 is trapped in the infrared radiations spectrum, the greenhouse effect would be enhanced because of the associated heating. In nature, sources of CO_2 in the atmosphere come from volcano eruption, carbonate rock weathering, decomposition of vegetation and other biomass, ocean release, naturally occurring wildfire, and respiration of living organisms. However, trends in atmospheric CO_2 show a remarkable increase, particularly after the industrial revolution (Callendar, 1938). Since then, CO_2 produced by human activities for living and manufacturing, including the combustion of fossil fuels, coal, natural gas, and oil as well as the wide use of cement, materially increase the concentration of atmospheric carbon dioxide with unequivocal evidence (Callendar, 1938, Keeling, 1961, Masson-Delmotte et al., 2018). Undoubtedly, the production of CO_2 and its influence on ecosystems have received great attention within the scientific community. Based on the observations from climate stations at Mauna Loa in Hawaii and Antarctica, Keeling (1961) measured that mean CO_2 concentrations have risen from 315 ppm in 1958 to 380 ppm in 2008, and further to 400 ppm in 2018 (Black, 2013). Additionally, the annual rise of global CO_2 concentrations is approximately 20 ppm. This rate is ten times higher than any recorded CO_2 rise value during the past 800,000 years (Masson-Delmotte et al., 2018). Le Quéré et al. (2018) have categorized the carbon budget into five classes to quantify the anthropogenic CO_2 emissions and its redistribution over the hydrosphere, lithosphere and atmosphere. These five classes are: 1) carbon emissions from fossil fuel combustion and other energy/heat oxidization; 2) the emissions variation driven by land cover change with the human interference; 3) the division of the above two carbon emission origins; 4) the uptake of CO_2 by oceans; 5) sinking CO_2 in the lands. By utilizing different research models which involve the historical emissions records, Le Quéré et al. (2018) have estimated that CO_2 emissions from fossil fuels rose to 34 Gt y^{-1} (Gt: Gigatonnes= 10^9 tonnes; y^{-1}= per year) in 2017 from an average 11.4 Gt y^{-1} during 1960-1970; while the emissions caused by land use and change covering agriculture and forestry are steadily progressing, where it was 5.5 Gt y^{-1} in the 1960s and 2010s and a slightly lower value (4.4-5.1 Gt y^{-1}) from 1970 to 2010. Overall, total CO_2 emissions due to anthropogenic activities over the period 1960-2017 have experienced a 140% increase. Meanwhile, the enhanced CO_2 sink function of terrestrial and ocean all have expressed clearly: the rate of capture of CO_2 has climbed from 4.4 to 11.7 Gt y^{-1} across the Earth's lands and from 3.7 to 8.8 Gt y^{-1} in oceanic systems, respectively (Le Quéré et al., 2018, Jia et al., 2019).

2.2.4 Carbon cycle

The element carbon, bonded with other elements, is found in nature as various chemicals in the whole carbon cycle. Mostly, the flux in the carbon cycle is presented as CO_2, which is more straightforward to measure as well as to react with other substances or to be formed during the biogeochemical process. The carbon fluxes as the form of CO_2 are described in Figure 2. 3, which shows that numerous mechanisms drive the fluctuation and variability of the global carbon cycle. The three dominant controlling factors that impact on carbon flux can be classified as: 1) short-term (biophysical); 2) middle-term (biogeochemical); 3) long-term (biogeographical) (Kondratev et al., 2003). Much research has been conducted concerning carbon flux in relation to its movement and interaction in different ecosystems.

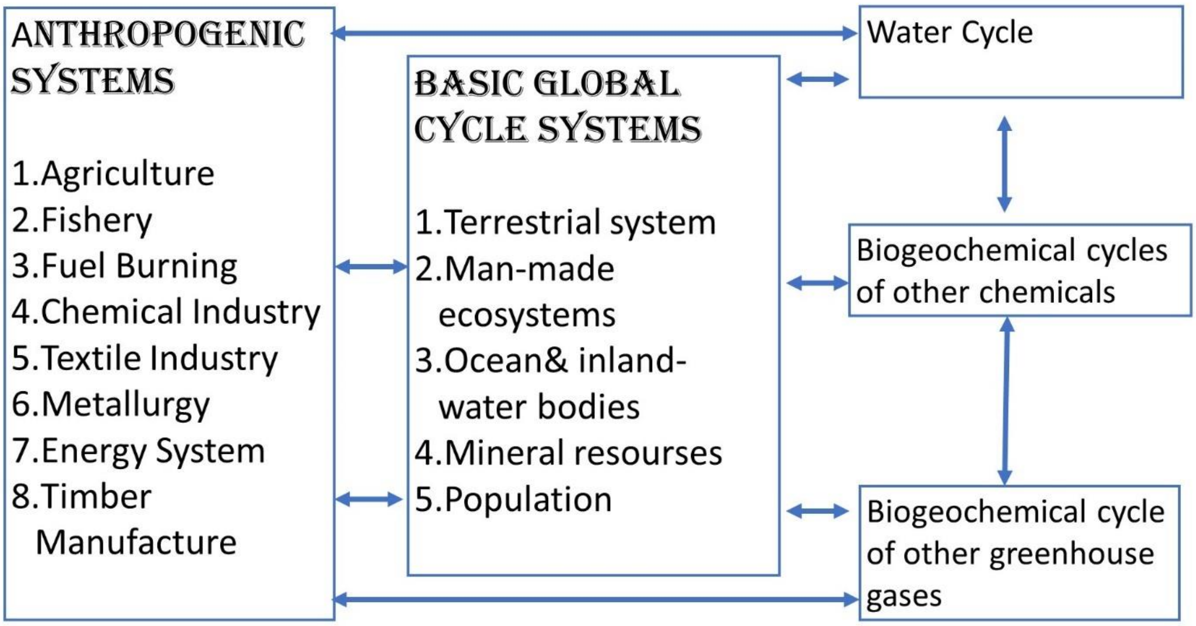

Figure 2. 3 The conceptual scheme of the carbon cycle in the environment. (Kondratev et al., 2003). (Adapted by Jiaqian Wang)

The carbon cycle, taking place at the point where atmosphere meets the ocean, is determined by the concentration of CO_2 as well as the gradient partial pressure of CO_2, temperature (solubility effect), and other hydrodynamic and hydrochemical parameters of the water domain. CO_2 either dissolves in the ocean or transports from the surface to deep layers due to the bioproductivity process and photosynthesis of phytoplankton, giving carbonic acid to the marine ecosystem (Kondratev et al., 2003). The land-climate process is an interactive consequence in terms of the exchange of GHG. Ciais et al. (2013) have established that a net sink of CO_2 by the whole of terrestrial ecosystems is comparable with the atmosphere-to-land flux of CO_2, if the land change-induced net carbon emissions are also involved in the modelling. Terrestrial carbon is conducted by three interactions:1. plant respiration and photosynthesis; 2. the decomposition of organic matter by microbes in soils; 3. disintegration and assembly of inorganic carbon based on the parent material and other substances during weathering and precipitation. All of these are also referred as “biogeochemical interactions” (Jia et al., 2019). In other words, the key factor influencing the assimilation of CO_2 in the land biosphere is net biome production, which is a balance between net primary production and carbon loss caused by heterotrophic organisms and burning of biomass in fields (Kondratev et al., 2003).

In terms of monitoring the global carbon cycle, the most common approaches applied are satellite observations and observations on-site (Kondratev et al., 2003). Due to the frequent and intensive intervention of human activities, especially fossil fuel consumption and altering one ecosystem to another one for different purposes, the variables of a functioning global carbon cycle are more complicated and vary with the spatial-temporal level. For example, the discharge of wastewater with abundant nitrogen from agricultural treatment and the iron deposition in rivers from dust generated by engineering constructions will enter the ocean eventually, which may facilitate a considerably greater CO_2 sink in the ocean (Ciais et al., 2013). Apart from satellite observations effectively supporting carbon cycle research, continuously updated mathematical models are necessities in demonstrating the evolution of carbon sources and the uncertainty of gas exchange. The computational models have greatly improved due to the advancement in the revised data on the land conversion rates, more observations in the supportive database and the simulation of air-sea flux variability in the open ocean. Especially since 2007, a new global compilation of forest inventory has substantially improved carbon cycle modelling (Ciais et al., 2013). Le Quéré et al. (2018) have summarised the change of annual global carbon budget from 2008 -2017 owing to human activities. For this time period, the summarized contribution of each principal component to total emissions is: 87% from fossil fuels combustion versus 13% from land-use change. Alternatively, the total emissions can be divided among the different Earth surface systems: atmospheric (44%) > ocean (22%) > land (29%). The remaining 5% accounts for the unattributed budget fluctuation (Le Quéré et al., 2018). In addition, from the national perspective, over the decade of 2008-2017, annual carbon emissions have decreased by 1.8%

across the 28 member states of the European Union, and by 0.9% in the USA. Conversely, an annual 3% increase in carbon emissions has happened in China (Le Quéré et al., 2018).

2.3 Terrestrial Carbon

The terrestrial system is an interactive and comprehensive environment where soil, organic organisms and living or dead biomass can be decomposed or aggregated by each other. This natural development can easily be positively or negatively affected by anthropogenic activities (Watson et al., 2000). Terrestrial C sequestration is achieved through soil organic/inorganic matter stabilization by physical occlusion within aggregates, chemical interactions with clay minerals, and biochemical recalcitrance (Ciais et al., 2013). Given all the complex features associated with soil carbon sequestration, the evaluation of soil condition, and the measurement of soil carbon stocks as well as estimation have been carried out by multiple research disciplines (IPCC, 2006, Le Quéré et al., 2018, Jia et al., 2019). Concerning C capture, the function of the global terrestrial sink, including organic and inorganic carbon, was 3.2± 0.7 Gt C· yr^{-1} during 2008-2017, increasing from 1.2± 0.5 Gt C· yr^{-1} in the 1960s (Le Quéré et al., 2018). In the earth's biospheres, soil organic carbon (SOC) pool (0-1m depth) contains the C of 1550 Gt (Lal, 2003). Furthermore, soil accumulates more than 3/4 of terrestrial organic carbon (Nele, 2013). Table 2. 1 displays the estimated capacity of the SOC pool among various land ecosystems. Globally, soil organic carbon was estimated around 1462-1548 Gt over the 1m soil layer (Batjes, 1996). Apart from soil organic carbon presented in Table 2.1, soil inorganic carbon (SIC), estimated at 695 to 748 Gt (0-1m depth), including lithogenic inorganic carbon and pedogenic carbon, significantly contributes to the carbon transition and carbon sequestration (Batjes, 1996, Lal, 2003). With respect to mitigating climate change, detecting, accounting and augmenting the amount of carbon in soil is a readily implementable option, which also supports the analysis about the relationship between soil carbon stock and GHG characteristics (Bangsund and Larry Leistritz, 2008, Whitehead et al., 2018).

Land ecosystems have been constantly and varyingly managed and evolved to meet a variety of requirements in human society, not just deforestation and agriculture cultivation, but also sea reclamation as well as urban infrastructure rebuilding. For instance, the amount of carbon emissions in the atmosphere resulting from land use change can account for 20% of total carbon emissions from anthropogenic activities (Ciais et al., 2013, Lal, 2003). Despite emitting a substantial amount of carbon, the land ecosystem can play a vital role as carbon reservoirs. Under the increasing land covers transitions with urbanization, the maintenance and optimization of soil functions such as transporting nutrients, regulating regional climate, providing food sources, outdoor recreational interests, and decreasing threats like serious degradation, are essential. The lack of high-quality residential environment, or surrounding by “dirty lands”, would jeopardize the well-being and health outcome of all living bodies. For example, work by Bambra et al. (2014) has shown that increasing proportion of brownfield land would significantly impact the health level and illness classifications amongst citizens.

Ecosystem	Area (10^9 ha)	SOC pool (Gt C)
Forests		
• Tropical	1.76	213-216
• Temperate	1.04	100-153
• Boreal	1.37	338-471
Tropical savannas and grasslands	2.25	247-264
Temperate grasslands and scrub land	1.25	176-295
Tundra	0.95	115-121
Desert and semi-desert	4.55	159-191
Cropland	1.60	128-165
Wetlands	0.35	225

Table 2. 1 Estimates of soil organic carbon (SOC) pool (Lal, 2003).

2.3.1 Soil organic carbon and inorganic carbon

The soil system attracts extensive attention from diverse research fields, not only because of its ecological impact on culture and technology but also on biomass production, energy transportation between organisms, carbon sequestration, creature habitats, the maintenance of a balanced ecosystem, etc. (Tang et al., 2018, Zheng and Han, 2018, Wiesmeier et al., 2019). Soil organic matter and soil carbon are the two main indexes in researching the mechanisms of carbon variation in the landscape (Banwart et al., 2015). Eroding by water and wind, geological parent material is disintegrated into fragments where infiltrating precipitation occurs and then elements or compounds are assimilated by photosynthesizing organisms. During this process, biomass such as plants and symbiont algae in lichen will intake inorganic carbon as CO_2. This is the dominant process by which soil organic C is formed, with soil microbes playing a major role in the transformation of organic C. The organic entities in soils always combine elemental C with other nutrients, such as nitrogen (N), potassium (P), and sulphur (S), in different ratios (Powlson et al., 2015), providing energy to soil organisms and plants. Conversely, from an organic-inorganic perspective, humic materials which are the colloidal product of advanced decomposition of biomass by soil organisms, can chemically bind to soil aggregates, which is the way clay mineral aggregates are formed (Banwart et al., 2015). By directly interacting with carboxyl or phenolic groups, organic carbon can work with clay minerals to form microaggregates which will further combine to form macroaggregates. Throughout the entire nutrients cycle, cations such as calcium (Ca), magnesium (Mg), zinc (Zn), and iron (Fe), will be exchanged and utilized, and this process is also imperative for maintaining the healthy growth of plants and complete soil structures (Powlson et al., 2015). Given the multiple reactions occurring in soils, the quantity of C fluctuates and other accompanied nutrients will also experience vast changes as well, e.g. in order to store 10,000

Kg C in humus, it is estimated that 833 Kg of N, 200 Kg of P and 143 Kg of S would be required (Lal, 2003).

Being an important constituent in soils and especially carbonate sedimentary rocks (Kondratev et al., 2003, Lal, 2003), terrestrial inorganic carbon can precipitate by the biogeochemical process involving atmospheric CO_2 and Ca $^{2+}$, Mg $^{2+}$, or other cations derived from soil silicate and carbonate minerals, which will occur over a considerable time scale (IPCC, 2006, Dong et al., 2019, Mercedes-Martín et al., 2021, Washbourne et al., 2015). Therefore, under natural conditions, the variation of the soil inorganic carbon pool led by land -use changes is slower because of a long geological time scale, which is not easy to account for soil organic carbon. Nevertheless, there are still many projects attempting to monitor the accumulation of terrestrial inorganic carbon under different circumstances such as the air-water interface, fertilization application and afforestation (Dong et al., 2019, Jia et al., 2019, Kondratev et al., 2003, Mercedes-Martín et al., 2021, Washbourne et al., 2015). Much research has been conducted in arid and semiarid areas, where its terrestrial inorganic carbon reservoir sink is generally 2-10 times larger than SOC (Schlesinger, 1982). In addition, considering the amount of massive demolition waste caused by increasing urbanization across the world, carbon stabilized in urban artificial silicates would present a great potential in soil inorganic carbon sequestration (Washbourne et al., 2015). Steel slag and red mud, for example, are two by-products from steel and aluminium industries, which could be utilized in carbon sequestration (Gomes et al., 2019). In the review of Gomes et al. (2019), the worldwide carbonation attributed to red mud fields is 572 Mt CO_2 (Megatonne; 1 Megatonne $=10^6$ tonnes), which is over 3.4 times higher than that of steel-making slag repositories. Together both alkaline wastes could sequester 3-4% of the global annual CO_2 emissions, offering a significant opportunity for these slag and red mud disposal areas to benefit carbon capture.

Several factors account for the turnover of carbon between organic and inorganic constituents. These include soil temperature, pH, particle characteristics and distribution in site (Canedoli et al., 2020), tillage frequency (Badagliacca et al., 2018), clay fraction (Chen et al., 2018), nutrient concentrations (Sarker et al., 2018), microbial diversity, climate features such as weathering strength and rainfall (Wiesmeier et al., 2019, Teh et al., 2011), etc. Consequently, soil formation is the result of complex interactions among organisms covering humans and animals, parent material, soil development chronosequence, topography, and regional climate, demonstrating that the study of soil carbon behaviour needs interdisciplinary approaches

(Goldhaber and Banwart, 2015). IPCC (2006) proposed several equations to compute the soil carbon stocks and also determined default values of carbon stock change caused by the different land conversions (Table 2.2). This report (IPCC, 2006) demonstrated a great amount of terrestrial carbon accumulation or loss has resulted from the alteration of land use, through the classifications of eight climate zones, seven soil types, twenty ecological zones and seven management practices over the world. For example, one principal equation for estimating soil organic carbon change is shown as (IPCC, 2006):

$$SOC = SOC_{REF} \times F_{LU} \times F_{MG} \times F_I \times A \qquad \textit{Equation 1}$$

SOC: soil organic carbon stock (tonnes C)

SOC_{REF}: the reference soil carbon storage (tonnes C/ha)

F_{LU}: stock change factor for land-use systems with the usage aim, dimensionless

F_{MG}: stock change factor for management regime, dimensionless

F_I: stock change factor for input of organic matter, dimensionless

A: the area of study site (hectare)

Apart from the scale of study areas, the other four independent factors of Equation 1 would change as land-use type changes. A simplified example of these calculations, such as cropland remaining as cropland but with various management methods, or other land use being transferred to cropland, can be found in Table 2. 2. As a result of land-use change, the final quantity of the soil carbon stock would show a substantial difference, implying that the net effect of soil carbon sequestration is the combined consequence of climate region, soil textures, spatial variability, and vegetation types. Among all diverse land conversion options, soils would accumulate more carbon in the wet and warm environment than the dry and cool environment, and more carbon in coniferous forestry as compared to broadleaved (IPCC, 2006). Objectives 1&2 of this study address the feature of soils in the farms and urban green spaces in north-eastern England and evaluate their carbon stocks under various land covers.

Factor	Level	Climate Region		Value
● Default reference soil organic carbon for mineral soils (tonnes C ha^{-1})				
SOC_{REF}	Lightly to moderately weathered soils	Cold temperate	Dry	50
			Moist	95
	Highly weathered soils	Tropical	Dry	38
			Moist	65

Factor	Level	Climate Region		Value
● Cropland remaining as cropland: Relative stock change factors for different management activities on cropland				
F_{LU}	Long-term cultivated	Temperate/Boreal	Dry	0.8
			Moist	0.69
		Tropical	Dry	0.58
			Moist/Wet	0.48
F_{MG}	No-till	Temperate/Boreal	Dry	1.1
			Moist	1.15
		Tropical	Dry	1.17
			Moist/Wet	1.22
F_I	High without manure	Temperate/Boreal/Tropical	Dry	1.04
			Moist/Wet	1.11

Factor	Level	Climate Region	Value
● Land converted to cropland			
F_{LU}	Native forest or grassland	All	1
	Shifting cultivation-shortened fallow	Tropical	0.64
	Shifting cultivation-mature fallow	Tropical	0.8
F_{LU}&F_{MG}&F_I	Managed forest	All	1
F_{LU}&F_{MG}&F_I	Moderately degraded grassland	Temperate/Boreal	0.95
		Tropical	0.97
F_I	Medium	All	1
	High	All	1.11

Table 2. 2 A small number of examples for factors of SOC_{REF}, F_{LU}, F_{MG}, and F_I used in Equation 1.
Noted that the latter three values are dimensionless, all of which are presented here for the cropland situation. Data sources come from IPCC (2006).

2.3.2 Carbon in vegetation

The significance of plants in soil weathering and planetary-scale evolution of the Earth has been acknowledged since early 1874 (Goldhaber and Banwart, 2015). As Table 2. 3 shows, Watson et al. (2000) have described the results of global carbon stocks from soil and vegetation, respectively, across various land use types and latitudes. Overall, the average carbon stock contributed by vegetation accounts for 19% of global carbon stocks, and 81% soil carbon pool. Carbon intake by plants by photosynthesis (see chemical equation below) is eight times higher than the carbon emitted across the anthropogenic activities per year (Joseph and Lehmann, 2009). In this process, carbon is fixed from gaseous CO_2 to a relatively stable form in sugars (the carbohydrates glucose and starch), as the energy source for vegetation growth. Therefore, an appeal to mitigate climate change by planting is prevailing, leading to substantially novel and cross-field studies about plant growth conditions under deforestation, afforestation or green land reconstruction, etc. (Kondratev et al., 2003).

$$6CO_2+6H_2O+photons=C_6H_{12}O_6+6O_2 \qquad \textit{Equation 2}$$

$$(\text{carbon dioxide } + \text{water} + \text{ light energy } \rightarrow \text{glucose } + \text{ oxygen })$$

Apart from an acknowledged carbon capture capacity, trees have a high potential in cooling the local environment by mediating the energy and heat exchange between ecosystems and atmosphere (Shukla et al., 2019), reducing flooding, and providing food and habitats for animals (Raum et al., 2019). Furthermore, the cultural impacts of trees on community resilience and social connections is vital for humankind’s well-being (Hand et al., 2019, Nowak et al., 2013).

Biome	Area (10^9 ha)	Global Carbon stocks (Gt C)		
		Vegetation	Soil	Total
Tropical forests	1.76	212	216	428
Temperate forests	1.04	59	100	159
Boreal forests	1.37	88	471	559
Tropical savannas	2.25	66	264	330
Temperate grasslands	1.25	9	295	304
Deserts and semideserts	4.55	8	191	199
Tundra	0.95	6	121	127
Wetlands	0.35	15	225	240
Croplands	1.60	3	128	131
Total	15.12	466	2,011	2,477

Table 2. 3 Global stock in vegetation and soil carbon pools (0-1 m depth).(Watson et al., 2000)

2.3.3 Canopy cover and carbon calculation of vegetation

The significance of trees on sequestering carbon increasingly attracts attention globally, and the ability to capture carbon in a single standing tree differs with the different tree components such as stem, branch or roots and particularly maturity stages (Aguaron and McPherson, 2012, Albert et al., 2014, Jenkins et al., 2003, Hale et al., 2019). Many scientists have progressed an amount of modelling and *in situ* data collection to calculate the carbon stock in vegetation across the regional or national scale (Albert et al., 2014, Kaplan et al., 2012, Jenkins et al., 2003, Jenkins et al., 2018, Nowak et al., 2013, Nowak and Crane, 2002). The Forest Services section of the United States Department of Agriculture (USDA) has developed multiple tools to determine the carbon stock for different purposes (https://www.nrs.fs.fed.us/carbon/tools/), such as PRESTO (an online tool to estimate the carbon owned by harvested timber), Carbon Online Estimator (COLE), i-Tree, U.S. Forest Carbon Calculation Tool (CCT) and a spreadsheet tool named “CVal” which enables land managers to evaluate the marketing value and profit potential of their green estate. Jenkins et al. (2018) have designed a comprehensive carbon assessment protocol for British forests. In Great Britain (GB), the National Forest Inventory (NFI) programme regularly surveys trees and other vegetation across GB every five years (https://www.forestresearch.gov.uk/). At the same time, other carbon assessment methods have been created such as CARBINE and C-FLOW (Hallsworth and Thomson, 2017). Likewise, the UK Woodland Carbon Code provides the online Carbon Calculation spreadsheet to enable people to calculate the carbon sequestered in their planting projects (https://www.woodlandcarboncode.org.uk/).

Allometric biomass equations, which calculate the tree carbon stock budget, need input physical parameters such as diameter at breast height (DBH) and height, and some need users to define tree species as well to obtain a more accurate estimation. A wide range of evidence has confirmed convincing results of applying DBH to establish the carbon stock (Nowak and Crane, 2002). DBH is the most common independent factor to calculate the vegetation biomass and further multiplying the biomass with a conversion factor to obtain carbon value. This conversion factor normally varies around 0.5 due to the varying growth rates and stages among different tree species. Again, the generalized regression relationship between biomass and DBH or height is an exponential function, which would slightly change with more or fewer coefficients, given the growth condition of different tree species.

Aguaron and McPherson (2012) compared four methods of estimating carbon stock in a western USA urban forest site (640 trees), using the allometric equations, i-Tree Street, i-Tree Eco, and Tree Carbon Calculator (CTCC) of CUFR (the Center for Urban Forestry Research). Their computations illustrated a maximum 29 % difference in carbon sequestration among the four approaches. Moreover, the difference in carbon storage results by these four methods on the single tree species varied, where CTCC always gave a greater number than i-Tree tools, largely because of an 0.80 correction factor designed for open-grown trees in i-Tree software. Apart from the ease in computation brought about by these carbon tools, other factors related to environmental services such as air pollution reduction, water runoff reduction, and the financial returns of tree-planting schemes, have been included in the advanced carbon calculator software (Hand et al., 2019, Raum et al., 2019). Raum et al. (2019) conducted an interview survey of feedback about the usage of i-Tree Eco (one tool from the i-Tree package) from 51 interviewees across Great Britain, from diverse backgrounds, including local councils, national park authorities and third sector organisations, etc. According to responses, the main barriers regarding the use of i-Tree Eco were not only questioning its few endorsements from the national commissions/governments, the lack of engagement across different policy, but also the drawbacks of the operation system linked to the i-Tree project delivery team (Raum et al., 2019). However, the identification of the ecological value of a tree, the suggestions for tree placement (Hand et al., 2019), and the assessment of woodlands or forestry composition in regions all enhanced the popularity of i-Tree Eco among diverse stakeholders (Raum et al., 2019). Unlike i-Tree Eco, which mainly examines carbon sequestration, woodland structure and replacements costs of the forest, another important i-Tree tool: i-Tree Canopy (which can estimate the canopy cover through aerial images), has been tested globally and positively accepted (Brent, 2014, Rogers and Jaluzot, 2015). By using i-Tree Canopy, 139 local governments across Australia estimated the land surface cover of their estates, demonstrating that totally grass-bare ground was the dominated land cover (47%) followed by tree canopy (39%) (Brent, 2014). Dozens of publications have shown that i-Tree packages provide a tailored assessment outcome for local ecosystem services (Brent, 2014, Hand et al., 2019, Rogers and Jaluzot, 2015) so Chapters 3&5 of this thesis will apply i-Tree Eco and i-Tree Canopy as well.

2.4 Reduction approaches for carbon emissions

In general, the approaches for reducing carbon emissions, enhancing carbon storage or sequestration can be categorized into three fields: biological, chemical, and physical (Figure 2. 4) (Shepherd, 2009). The development of carbon removal should be compatible with newly advanced technology, involving more human participation and integrated design, because the natural process of reducing carbon emissions such as geological weathering or ocean fertilization is quite slow, which will take a very long time to make an effect on carbon sequestration (Shepherd, 2009, Masson-Delmotte et al., 2018). Mitigation pathways for carbon emissions cover many aspects, including the utilization of new energies (bioenergy, hydrogen &solar power), low-carbon infrastructure improvement (the smart grid system on energy transportation, allocation and monitoring), the innovation of traditional biomass usage ways (soil amendments and biofuels), green finance and industries (waste recycling, carbon footprint assessment), the establishment of carbon pricing and trading markets, land resources-based mitigation options such as dietary change, manure management and land use type conversion which includes “natural climate solutions” such as forest and wetland management (Cornell University, 2013, Dong et al., 2019, Hansen et al., 2012, IPCC, 2006, Jia et al., 2019, Wang and Sainju, 2014). Due to the “Net-Zero Carbon” target from different countries or policy domains, lots of institutions proposed their own internal carbon management plans. The UK government has published a series of Carbon Budgets for the aim of “Net Zero” by 2050. In December 2020, the 6th Carbon Budget has been announced, which will be put in place from 2033 to 2037, and proposed a great number of pathways to provide alternative contributions of carbon abatement (Climate Change Committee, 2020). In the UK, universities have taken action on reducing carbon emissions because the Higher Education Funding Council for England (HEFCE) and a representative body for UK Higher Education named GuildHE, together, in 2010, announced their intention to achieve a 43% carbon reduction by 2020, against a 2005 baseline (Lewis and Patton, 2010). The published carbon management plan (CMP) containing a step-by-step institutional carbon reduction target and aligned carbon removal approaches, from sixteen universities (12 in the UK and 4 beyond UK), is included in **Appendix A**. Basically, most universities attempt to mitigate carbon emissions from behavioural change, green transportation and IT, energy saving, building refurbishment, waste recycling, monitoring and surveying, space utilisation and cooperation, etc. With regard to optimizing land management, this was suggested by only two universities. These considered maintaining the biodiversity, reducing the threat from exotic plants (The University of Tasmania) and exploring the possibility of cropping and biofuels in the available ground (State University of New York at Buffalo) (**Appendix A**).

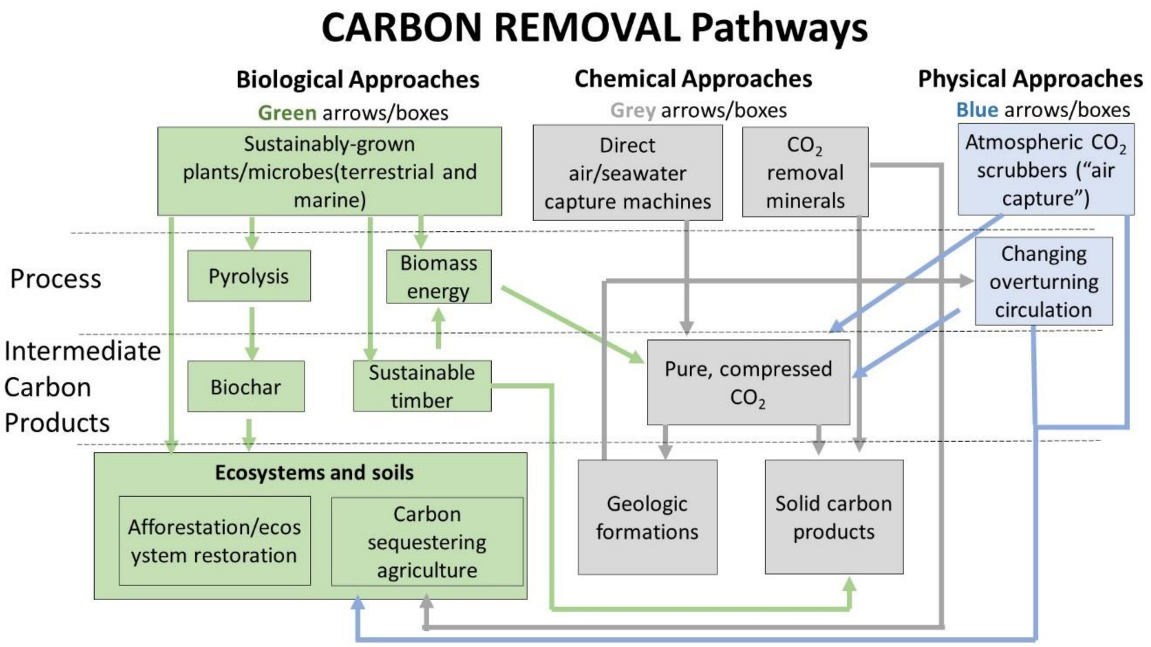

Figure 2. 4 Three carbon removal pathways and how do they relate to each other. Sources from (Noah, 2015, Shepherd, 2009) (adapted by Jiaqian Wang).

2.5 The function of land cover conversion on carbon emissions and sequestration

Land use could bring about substantial carbon sequestration (Bangsund and Larry Leistritz, 2008, Kaplan et al., 2012, Rees et al., 2018, Wang et al., 2019, Wiesmeier et al., 2019). After altering land cover, the exchange of hydrological mass, aerosols (dust, carbonaceous particles, etc) and gases in the land would all change concurrently, resulting in variation in water quality, temperature, and the strength of horizontal and vertical winds, which in turn influence the concentration of CO_2 in the atmosphere (Jia et al., 2019). The overwhelming majority of

Earth system models have been developed to monitor the change of carbon storage and emissions in the atmosphere caused by diverse land designs and management, with many trials embedded to the combined variation in both biophysical and biogeochemical processes (Le Quéré et al., 2018). Kaplan et al. (2012) collected the terrestrial dataset for Europe over the past five centuries. They found that anthropogenic land usage had the dominant effect on carbon storage compared with versus other factors such as the variability of climate and CO_2 concentrations. According to the results run by the Lund-Potsdam-Jena dynamic global vegetation model, Kaplan et al. (2012) found that the carbon stock in the European continent could probably have increased by 5.6% (217.1 Gt to 230 Gt) between 1500 to 2000 if the terrain of Europe was kept as natural vegetation, while the carbon stock showed a 15.6% decrease (202.1 Gt to 174.5 Gt) considering the human impact on the historical European land use during these five centuries. Similarly, the deficiency of land management practices would bring a detrimental effect in speeding the accumulation of carbon concentration in the atmosphere. In 2017, the average anthropogenic CO_2 emission caused by global land-use change was 1.4± 0.7 Gt C, which is similar with those of previous decades, although prediction lacks a high confidence (Le Quéré et al., 2018). The transformation of land cover also plays a vital role in soil carbon (Goldhaber and Banwart, 2015, Hansen et al., 2012, Wiesmeier et al., 2019). In the UK, Hallsworth and Thomson (2017) released the dataset describing how soil carbon sequestration is affected by land use change (Table 2. 4). In 2015, the substantial change in the soil carbon balance in the UK still predominantly resulted by land conversion to forestry and grassland (carbon sequestration) and to cropland (carbon emissions), while other land use activities, such as non-forest land converted to grassland or forest land converted to settlement, was causing a comparatively smaller fluctuation of carbon dynamics (Table 2. 4). Meanwhile, the UK 6th Carbon Budget has scaled up land conversion where a further 260,000 hectares of arable lands will be transferred to be bioenergy use; likewise, the area of woodland will rise to 15% by 2035, from the present level of 13% (Climate Change Committee, 2020). At the same time, the restoration of peat land is expected to increase from currently 19% to 58% in 2035 across the UK. Overall, the land-based carbon sinks could accumulate up to an estimated 23 Mt (million metric tonnes) CO_2 in 2035, which brings a 28% increase since now (Climate Change Committee, 2020).

Activity	**2015 UK total Gt C emissions (+) or removal (-)**	**Group**
Land converted to Forest land and land remaining Forest Land (not including emissions from wildfires)	-4347.79	Forest Land
Land converted to Cropland and land remaining Cropland (change in soil carbon not including losses from drainage of organic soils)	2835.53	Emissions from soils due to land use change
Land converted to Grassland and land remaining Grassland (change in soil carbon not including losses from drainage of organic soils)	-2691.47	Emissions from soils due to land use change
Land converted to Settlement and land remaining Settlement (change in soil carbon)	1620.25	Emissions from soils due to land use change
Cropland remaining Cropland (drainage of organic soils)	464.15	Emissions from soils due to drainage
Forest Land converted to Grassland (deforestation to grass – not including soil changes)	203.72	Minor emissions
Cropland remaining Cropland (cropland soil management practices)	-107.26	Minor emissions
Wetlands remaining Wetlands (peat extraction)	73.28	Minor emissions
Non-Forest land converted to Grassland (change in non-forest living biomass)	-62.48	Minor emissions
Grassland remaining Grassland (drainage of organic soils)	48.22	Minor emissions
Forest Land converted to Settlement (deforestation to settlement – not including soil changes)	45.25	Minor emissions
Non-Forest land converted to Cropland (change in non-forest living biomass)	36.03	Minor emissions
Cropland remaining Cropland (cropland biomass management practices)	-24.36	Minor emissions
Grassland remaining Grassland (grassland biomass management practices)	14.97	Minor emissions
Non-Forest land converted to Settlements (change in non-forest living biomass)	-14.52	Minor emissions

Table 2. 4 The carbon emissions and removals because of Land use, Land Use change and Forestry in the UK in 2015. (Hallsworth and Thomson, 2017)

2.5.1 Land use conversion in agriculture and woodlands

25% of global greenhouse gas (GHG) emissions is attributed to forestry, agriculture and other land uses, particularly including CO_2 emissions from deforestation, CH_4 emissions from rice and ruminant livestock, and N_2O emissions from fertilizer spread (Arneth et al., 2019). Given the rapid increase of GHG emissions from agriculture over the past decades, many studies have explored the responsibility of agricultural land, grassland and grazing land in capturing carbon from the atmosphere (Badagliacca et al., 2018, Bangsund and Larry Leistritz, 2008, Chen et al., 2018, Follett and Reed, 2010, Lal, 2003, Wang and Sainju, 2014). Agricultural soils can be a vital sink to sequester carbon (Jarecki and Lal, 2003, Masson-Delmotte et al., 2018), and the accumulation of soil carbon can provide various advantages for farming soil environments. First of all, for agricultural soils, primary crop production could be increased. The growth of farm products depends on nutrients which are regulated by mineralization or immobilization processes and in the same way, soil carbon is the substrate of soil microorganisms (Goldhaber and Banwart, 2015). In addition, the quality and quantity of water are influenced by soil carbon, because the percentage of SOC would change soil properties that control the moving path of water in soils (Powlson et al., 2015). Soil aggregates, which act as filters to absorb the pollutants from herbicide and pesticide residues, would increase with the increasing SOC. Moreover, soil is the largest terrestrial reserve of biodiversity, and its stability is threatened by soil erosion, salinization, acidification, soil contamination, decline in organic matter, prolonged cultivation, nutrient imbalance, waterlogging, loss of soil biodiversity. Soil carbon is a fundamental constituent of the biosphere and the increase of SOC is the driving force for a prosperous ecosystem (Jeffery and Gardi, 2010). In a previous study, C sequestration not only mitigates climate change but also enhances soil fertility and the productivity of ecosystems (Wang and Sainju, 2014, Wiesmeier et al., 2019).

Lal (2003) postulated that changes in emissions, chemical and physical reactions, and the stabilization rate of carbon in agroecosystems were the final consequences of anthropogenic activities. Therefore, one of the methods proposed by Lal (2003) to increase the contribution of agricultural lands in sequestering carbon is to implement management practices which have been recommended previously with a high degree of confidence. Crop rotation is one significant part of farm management practices, because the selection of previous crop would determine soil quality and structure, nutrients composition, the presence of pest, etc, and thereby influencing the productivity of the subsequent crops and the accumulation of soil carbon (Jarecki and Lal, 2003). Instead of farming different crops, the factors in farm planning which could significantly affect agricultural soil carbon include: the input of inorganic and organic fertilizers, the suitable irrigation within water management, the method of tillage, surface crop residue management and more, as explained by Lal (2001), Jarecki and Lal (2003), and Whitehead et al. (2018). Badagliacca et al. (2018) performed a 2-year study in Italy in order to determine the carbon difference between the application of no-tillage and conventional tillage where both farming practices have been conducted over 20 years, and they found that no-tillage could capture a more C stock of 0.70 Mg ha^{-1} $year^{-1}$ more than the conventional tillage. Sarker et al. (2018) suggested that the stock of carbon and nutrients in farm soil systems can be enhanced by reducing soil disturbance and applying crop residues incrementally. In contrast, Chen et al. (2018) concluded that carbon sequestration is still closely related to the species of crops and planting patterns, even under optimized farming treatments.

Jarecki and Lal (2003) thought the SOC pool would experience a loss under agricultural systems, but that it can reach an equilibrium in other ecosystems (e.g. forests or prairies). Conversely, poor management of land change and cultivation practices would cause more carbon emissions than the combustion of non-recyclable resources (Lal, 2003). As a consequence, a comprehensive research survey which involves assessing the balance of croplands and woodlands and the length of time that they would be retained as part of management practices, is essential (Chen et al., 2018). Then, because of the complication of an integrated management scheme containing agricultural lands and woodlands, the feasibility of converting management practices needs to be clarified (Wiesmeier et al., 2019). Furthermore, the expansion of a methodological framework for farms, from a single ecosystem to a more complex combination of mixed ecosystems, is urgently needed for a better understanding (Zheng and Han, 2018).

2.5.2 The role of urban open green areas on climate action

Over 50% of the global population lives in urban areas (Jia et al., 2019) and IPCC (2018) revealed that the CO_2 emissions from all cities globally are responsible for 70 % of the total worldwide amount. Another predicted 10% of the world's population will emerge into cities by 2030 (United Nations- Population Facts, 2020), indicating massive new urban lands required to be converted from rural areas in order to provide the housing space associated with fundamental urban structures (Jia et al., 2019). The classification of urban areas is more diverse than the natural environment due to anthropogenic needs, where one place can perform several functions in order to maximize its service efficiency in city. Plus, the related fossil and electricity energy consumptions, with carbon emissions, has considerably increased in urban areas since the Industrial Revolution. The vast amount of CO_2 emissions in modern cities, where the majority is from vehicles and building activities, is a serious issue that exaggerates climate change. This is difficult to alleviate because the increased energy use with associated CO_2 emissions has continued with the massive expansion of urbanisation (Pan et al., 2019, Duren and Miller, 2012). Björkegren and Grimmond (2018) calculated an annual total of 51.4-53.5 Kg CO_2 m^{-2} yr^{-1} in central London with respect to the storage and vertical advection of CO_2 from June 2012 to May 2013. The two largest municipalities in China (Beijing, Shanghai), caused a total of 426 MMT (million metric tonnes) of carbon emissions in 2012 based on the high-resolution emission gridded database (Liu and Cai, 2018). Large cities are facing demands to mitigate climate change at various levels depending on their abilities. Otherwise, a series of adverse consequences resulting from the worsening urban climate, i.e. the increasing soil salinity, landslides from precipitation extremes, the well-known urban heat island, would significantly threaten the amenity of the settlement environment and the function of the fundamental urban infrastructure (United Nations-Population Facts, 2020). Under abnormally high temperatures, the outcome of labour productivity is unfavourable (Jia et al., 2019). In Los Angeles, the city council set the goal to reduce GHG emissions by 35% below 1990 levels by 2030 (Los Angeles City Council, 2007); in the entire city of Sydney, there is a 44% carbon reduction target against the 2006 baseline by 2021, and the 70% reduction of GHG emissions by 2030, and achieving further net zero emissions by 2050 (City of Sydney News, 2020). Covering 885 cities across 28 European countries, where the population of the majority of sampled cities is over 50,000 while few urban areas with less than 50,000 inhabitants were also included, the project of Reckien et al. (2018) has categorized the local climate plan of these cities according to their engagement motivation and the impact led by the higher administrative policy bodies. They reported that around 67% of sampled cities developed a climate plan autonomously or compulsorily while the remaining 33% of cities have not made the decision-support mechanisms needed to the urban greening and sustainability (Reckien et al., 2018). Meanwhile, some cities have already cooperated to develop the global green new deal in order to cut carbon emissions and improve the urban dwelling environment. One fascinating example is C40 (2019) founded in 2005, connecting 96 of the world’s largest and most influential cities, and committing to integrate the contribution of each individual city to keep global heating below 1.5 ℃ and halving the GHG emissions by 2030. Given this kind of international participation and contribution, some major cities including London, New York and Paris have reduced the CO_2 emissions by an average of 2% each year after reaching the GHG emission peak in 2012 (Fingas, 2018). However, the emergence of metropolises is still producing an increase in dwelling density and the land resources that can be utilized are quite limited when designing or renewing the landscape in the urban areas (Jorat et al., 2020). Even though integrating the urban green infrastructure into city development is challenging, the outcome of urban green infrastructure is more effective than conventional infrastructure (Masson-Delmotte et al., 2018). For example, green streets, green roofs and green walls can alleviate the heat island effect; bioretention cells, bioswales and permeable pavements effectively slow and filter stormwater flows; a biodiverse urban habitat such as wetlands and riparian buffer zones can build urban resilience as well as being a recreational environment for citizens.

As a natural sink, urban trees significantly affect carbon cycles and dynamics with the filtration of air pollutants (Hale et al., 2019, Hallsworth and Thomson, 2017). Other highlighted contributions such as the expansion of complex tree roots augment urban hydraulic structures, absorption of runoff water, and nutrients storage (Raum et al., 2019, Albert et al., 2014), all prompting increasing research on urban trees (Pan et al., 2019). Nowak has made many estimates of carbon stock by urban trees in the USA: the data suggested that in 2002, carbon stock and annual gross carbon sequestration in the total of urban trees across the USA was 700 million tonnes and 22.8 million tonnes C yr^{-1}, respectively (Nowak and Crane, 2002). In 2013, these two values correspondingly changed to 643 million tonnes and 25.6 million tonnes, with 38.6% urban tree cover (Nowak et al., 2013). The UK government pledges a planting scheme in which 30,000 hectares of trees per annum across the UK would be achieved by 2025. As a part of that scheme, the Urban Tree Challenge Fund aims to plant 13, 400 new trees across cities and towns in England including more than 50,000 trees particularly for urban regions. In Leicester, UK, Edmondson et al. (2012) have demonstrated that 18% organic carbon across the city region is sequestered in urban trees: carbon storage averaged of 4 $Kg\cdot m^{-2}$ over the entire city, but varied from 1 $Kg\cdot m^{-2}$ in residential areas to around 5 $Kg\cdot m^{-2}$ in non-residential areas. Furthermore, appropriate tree species selection could improve the ecological service delivery and carbon storage of urban green space, reduce the damage to urban grey fundamental infrastructure due to root growth, and alleviate the maintenance burden of ground managers (Watson et al., 2000, Nowak et al., 2013). Concurrently, Lal and Augustin (2011) stated that the constraints of surrounding constructions or interrupted nutrient cycling in the city green land could hamper the lifespan, tolerance and shape of urban trees. They also found that Cedar and Larch trees possess the highest potential for carbon storage among 145 urban tree species in Chicago (Lal and Augustin, 2011). Urban deciduous trees are able to adjust the regional heat effect among intensive buildings, while coniferous trees perform well in alleviating energy emissions during the winter being ever-green and with bottom to top needle leaves which can block strong winds for houses (Lal and Augustin, 2011). Selecting a busy urban network (250 m ×200 m) in Newcastle upon Tyne in the UK, Tiwary et al. (2016) investigated seven performance indices from 11 tree species and 4 types of shrubs, where the performance indices include pollution flux potential, carbon sequestration potential, thermal comfort potential, noise attenuation potential, biomass energy potential, environmental stress tolerance and crown projection actor. According to their calculations, the carbon sequestration and biomass energy capability of London Plane and Willow could present the highest potential, while most of the vegetation species were less likely to alleviate the street noise effectively except Spruce and shrubs (Tiwary et al., 2016). By conducting i-Tree Eco and compiling the outcome of ten case studies in Great Britain, Hand et al. (2019) described the diversity of gross carbon sequestration and carbon storage for twelve tree species as trees mature, and found Oak *spp* in the mature age classification stored the greatest amount of carbon while the ability of Leyland Cypress was the least effective, and the authors acknowledged that i-Tree Eco is fit for purposes for urban ecosystem evaluation.

Aboveground carbon captured by urban vegetation is vital to mitigate regional climate change, but inputs of carbon into urban soil should be attached equal importance, especially considering the easy access of air pollutants into the soil due to increasing rapid urbanization and the concomitant anthropogenic activities (Renforth et al., 2011). The frequent and intensive human interventions disturb the original soil profile sequences and nutrients recycling approaches (Canedoli et al., 2020). For example, there might be abundant organic and black carbon in the subsoil horizons due to the substantial relocation of topsoil horizons during construction programmes (Lorenz and Kandeler, 2005). Also, the presence of compaction would decrease the carbon density in the soil because it hinders the carbon storage process from the fresh surface to the deeper soil profiles (Louwagie et al., 2016). Considering the pressure for solving the excessive carbon emissions in cities and the incomplete urban soil function nowadays, proposing feasible and economical schemes to maximise the potential of urban soils in remediating the carbon emissions is inevitable (C40, 2019). Jorat et al. (2020) think there would be a temporary time when the demolished area might be used as green space as well, either formally or informally, before new construction starts and thereby offering an opportunity for the extra photosynthetic carbon to stay in soils during this vacant period. Afterwards, carbon can be sealed in the urban soils during the later construction work. Besides, urban “grey” infrastructure, such as roads and supply pipelines underground, require earth works. Thus, excavations conducted on the pavement with different purposes facilitates the access to subsurface soil where the soil amendments, such as biochar, can be applied for improving the high-temperature performance of asphalt, attenuating heavy-metal contamination and sequestering carbon (Louwagie et al., 2016). In addition, sustainable drainage systems have been put in place to manage local surface water flooding in many regions, e.g. the sponge city construction program in China (Li et al., 2017), and the 2BG “Black, Blue & Green” project based on the sewer network in the catchment area in Denmark (Fryd et al., 2009). In Newcastle city centre, UK, a pilot swale has been constructed as a living laboratory, to monitor stormwater runoff and infiltration with the implementation of hydrometric sensors (Figure 2. 5). With increasing attention on adopting sustainable drainage systems in cities, the auxiliary functions of sequestering carbon into sustainable urban drainage networks should be carefully considered as well.

Figure 2. 5 Photos of Extreme Event Swale -a sustainable drainage system included in the National Green Infrastructure Facility of Newcastle University. (Image Jiaqian Wang)

2.6 Biochar technology

Biochar is the carbon-rich product obtained through a heating process with little or no oxygen. This material is usually produced from the biomass of wood, manure or leaves. The term “biochar” was initially proposed in the context of C sequestration in soils and as a long-term carbon sink (Lehmann et al., 2006); later, Joseph and Lehmann (2009) described biochar as the carbon-rich product obtained from biomass when heated at relatively low temperature (<700°C) with a limited supply of oxygen. In the latest Biochar Standards, the International Biochar Initiative (2015) explains that biochar is a solid material, converted from the biomass under a thermochemical and oxygen-limited environment, which aims to improve soil quality and resources efficiency, and to remediate the adverse effect of pollution and the climate change emergency. This new approach can manage materials that might otherwise be considered as waste while sequestering carbon, improving soil fertility and water-holding capacity, and providing the bioenergy which is a by-product of the pyrolysis process. Consequently, extensive research has taken place in terms of biochar production and application with its conditioner characteristics in the amended environment (Joseph and Lehmann, 2009). Biochar pyrolysis oxidizes 45-48% of the organic matter of parent materials to CO_2, while the natural burning of residues would emit almost 90% of C in the organic matter to the atmosphere, so the complete combustion during biochar pyrolysis can alleviate 420-450 Kg C emissions per tonne of C contained from feedstocks (Lehmann et al., 2006). Meanwhile, the bioenergy produced during the pyrolysis is potential green energy. Bioenergy can be utilized where fossil fuel consumption is expected to decrease, although this alternative is less common (Lehmann et al., 2006). Lehmann et al. (2006) have suggested that if the slash and char method were applied globally, instead of slash and fire, a total of 0.21 Gt C, accounting for 12% of the total C emissions in human activities would be offset per year. According to the current stock of feedstock and the conversion rate of raw material to biochar, Windeatt et al. (2014) have listed three scenarios about global carbon sequestration potential owing to long-term biochar treatment: under the maximum 100% residue availability scenario, the predicted long term carbon storage potential is 0.55 Gt CO_2 yr^{-1}, while this value is 0.28 Gt CO_2 yr^{-1} and 0.06 Gt CO_2 yr^{-1} for 50% and 10% of the residue availability, respectively. Furthermore, Windeatt et al. (2014) have proposed that, in the maximum 100% residue availability scenario, 93% of the entire carbon sequestration potential would be from the agricultural waste biomass in rice husk, wheat straw and sugarcane bagasse.

Moreover, soils amended by biochar can be an effective sink for carbonate precipitation, which is a critical contribution to sequester inorganic carbon (Masson-Delmotte et al., 2018). The mechanism of inorganic carbon formation can be expressed simply as below:

$$CaAl_2Si_2O_8 + CO_2 + 2H_2O \rightarrow CaCO_3 + Al_2Si_2O_5(OH)_4 \qquad Equation\ 3$$

Capturing CO_2 through the natural process via the carbonation of calcium and magnesium is slow, but biochar can provide a higher pH environment and electrical conductivity as well as Ca^{2+}, and hence accelerate the formation of soil inorganic carbon (Dong et al., 2019). Additionally, the multi-pore structure of biochar can harbour microbial biomass and thereby facilitating carbonate precipitation (Dong et al., 2019). Regarding the accumulation of inorganic carbon in urban soils using biochar where the parent material includes cement kiln dust, blast furnace slag, anthracite ash, steel making slag, etc, this maximum potential was approximately estimated as 7 Mt C yr^{-1} in the UK (Renforth et al., 2011).

2.6.1 Priming effect in soil due to the biochar amendments

As a soil amendment, the performance of biochar in mitigating climate change could be explained by the following points:

1) pyrolysis processing of waste biomass to biochar is the alternative method to re-use crop residues, and animal manures, which is beneficial to reduce the GHG emission burden of landfilling and crop residue combustion. During a couple of hours' pyrolysis, non-graphitic C and amorphous cracks will be replaced with graphitic C and crystallographic order in the three dimensions (Franklin, 1951);
2) the increase in surface area of biochar helps the accumulation of microbial biomass populations in original soils where only sand or clay exist (Downie et al., 2009); meanwhile, the pore diameter scanned in biochar macroporosity ranges from 5 μm to 10 μm, which it is a desired incubation environment for microbial cells like bacteria, fungi and lichens. These microbial organisms' sizes typically ranges from 0.5 μm to 5 μm, and algae which size is in the range of 2 μm- 20 μm (Downie et al., 2009). The abundance of arbuscular mycorrhizal fungi which is correlated with soil aggregate enables more C accumulation in multiple ways. One that may be of particular interest in the context of biochemical mechanisms is enrichment in the protein glomalin (Thies and Rillig, 2009).
3) C mineralization rate would be slower, partially because CO_2 can be chemisorbed at biochar surfaces, especially when applying high temperature biochar (Nele, 2013) and partially owing to the increased anion/cation exchange capacity characteristic of biochar, which improves the absorption of nitrate and ammonium and thus causes N shortage (Nele, 2013).
4) the increase of pH due to negative charge on the biochar surface buffers acidity in soils (Joseph and Lehmann, 2009), accompanied with the supply of ash containing

abundant Ca^{2+}, which may benefit the formation of soil pedogenic carbonate (Dong et al., 2019).

However, a great variety and uncertainty of the priming effect of biochar on soil carbon sequestration has raised lots of arguments. The effect of soil carbon sequestration after biochar deployment varies with the different feedstock types, pyrolysis conditions such as the temperature of pyrolysis and the hold time under the maximum temperate, and application rate in the fields (Dong et al., 2019, Yang et al., 2019, Windeatt et al., 2014). Nele (2013) confirms that the volatile matter content of willow wood biochar, which facilitates the formation of a microbial substrate, is a significant factor leading the cumulative soil CO_2 emission in a short-term lab experiment, while soil amended with low temperature (300℃) biochar emitted more CO_2 as well as having a higher microbial activity than biochar produce at high temperature (750℃). Similar results are reported from Yang et al. (2019), who study the difference of carbon balance and leachate quantities between two treatments: one with the biochar made from mixed pine tree and larch and the other from sewage sludge. They found that the largest amount of CO_2-C loss occurred in woody biochar at low temperature and that sewage sludge biochar negatively impacts CO_2 production (Yang et al., 2019). At the same time, by monitoring four multi-year trial sites with various woody biochar mixtures applied at different rates (one site in the UK and three sites in Italy), Nele (2013) has found that only the newly established cultivation field and the field with the highest biochar proportion could show an apparently greater SOC, but C mineralization rates are relatively lower (the negative priming effect) in biochar-cultivated soils compared to their corresponding control groups. In addition, unlike the short term experiment, microbial activity and biomass in these four long-term field trials all decreased compared to the control plots and hence Nele (2013) has stated that the 1-4 years aging biochar, as a substrate, would not function as strongly as at the start. Kuppusamy et al. (2016) stated that the reduction in soil albedo caused by biochar addition would bring a detrimental effect in carbon abatement. Conversely, the experiment of Zhang et al. (2018) indicated that even though the soil surface albedo would reduce by 21-45% with various chestnut wood biochar amendments rates, CO_2 efflux in soils was not influenced by this physical change.

Additionally, how the stability and longevity of biochar in soils would develop has yet to be fully understood (Kuppusamy et al., 2016, Nele, 2013, Shackley et al., 2010). Of all challenges about assessing the stability and potential of biochar, three factors including the mean residence time, carbon stability index and how to monitor biochar in situ are the hardest to solve (Shackley et al., 2010). In order to calculate the recalcitrance potential of biochar, Windeatt et al. (2014) used the R_{50} values (a recalcitrance index) to classify the degradability of several biomasses. The sequence ranking the most degradable biomass to the least is: wheat straw, coconut fibre, cotton stalk, sugarcane bagasse, rice husk, olive pomace, and palm shell. Lignin content in the biomass may account for this (Windeatt et al., 2014). Aging biochar cannot be supportive as a fungal substrate during metabolic activity or be a rich source with N, which would hinder the C mineralization rate (Nele, 2013). Apparently, in the environment amended with biochar, the diminishing rate of positive effect on C mineralization among soil organic matter is unlikely to be linear (Shackley et al., 2010). In summary, the impact of biochar amendments on soil carbon sequestration to mitigate climate change is complex (Kuppusamy et al., 2016).

2.6.2 Effects of biochar on nutrient leaching

Leaching causes nutrients loss in soils because of rainfall or surface run-off. Also, the leaching of metals as well as organic compounds potentially severely pollutes the ground water, which has been a worldwide threat to the safety of agriculture irrigation and drinking water. In addition, the abundant accumulation of phosphorus and other nutrients in surface water leads to eutrophication, which would damage aquatic biodiversity and water quality (Joseph and Lehmann, 2009). These environmental issues can find a solution in biochar usage. Along with the high cation exchange conductivity (CEC) possessed by biochar, the greater sorption appearing as the high porosity of biochar and increased water holding capability in the biochar-mixed soils are able to remediate the leaching of toxic cation contaminants, and thereby protect the ground water from heavy metal pollution (Joseph and Lehmann, 2009, Kuppusamy et al., 2016). Plus, based on the biota perspective, the mixture of soils and biochar benefits mycorrhizal communities, which creates the ability for plants to intake nutrients, and whereby the loss of microelements is prevented (Joseph and Lehmann, 2009). Data of Yang et al. (2019) have illustrated that sewage sludge biochar effectively remediates the concentration of chemical oxygen demand (COD) in the leachate while both sewage sludge and woody biochar from 700℃ significantly stock total nitrogen in soils. Again, biochar processed from these two feedstocks, although at two different pyrolysis temperatures, can ameliorate the dissolved carbon in the leachate. The remarkable performance of sewage sludge biochar in decreasing the leaching of COD partially was attributed to the increased presence of functional groups carrying O in biochar (Yang et al., 2019). However, the magnitude of bonding cations or anions differs with the variability of biochar characteristics, so the intake or sorption of micronutrients from the soil-biochar matrix may vary (Kuppusamy et al., 2016), possibly accounting for adverse effects on plant growth, if seen.

2.6.3 Assessment systems and implementation challenges related to biochar technology

The biochar industry has been developed in many countries and its uses and benefits have become better known. For example, The UK Biochar Research Centre (UKBRC) has been regulated by and cooperates with the United Kingdom Department for Environment, Food and Rural Affairs (Defra), and the Department of Energy and Climate Change (DECC) to examine the deployment of biochar in the UK. The UKBRC not only reviews the feedstock availability, environmental behaviour of biochar but also covers the associated ecological benefits and disadvantages, and develops the life cycle assessment connected with an estimation of pyrolysis investment at different scales, and revenue from various timeframes (Shackley et al., 2010). Importantly, establishing the carbon balance baseline is essential, especially when a large amount of fossil-fuel is burned during biochar production, because it is not sustainable if CO_2 produced in the pyrolysis is more than the carbon stored in the biochar (Kuppusamy et al., 2016). In addition, based on the four criteria regarding effectiveness, timeliness, safety, and cost, Shepherd (2009) has evaluated the application of biochar compared to other carbon removal approaches. As a result, the effectiveness of biochar would be limited by parent material yield. The potential conflicts on land use for farming or bioenergy is less possible to solve in the short term, considering that encouraging of biomass C sequestration could increase the cost of crop foods (Shepherd, 2009). There are several difficulties in popularizing the biochar application, particularly the economic and logistical challenges, its mean residence time, stability and the ease of monitoring its variation in situ (Shackley et al., 2010). Moreover, the doubts about biochar application are constantly being argued in the following areas:

1) Resource requirements: not all organic wastes processed in biochar pyrolysis units are easily collected in agricultural activities, and thus the available high-functionality of organic feedstock would be particularly competitive, which raises the market price (Shackley et al., 2010). In addition, the design of pyrolysis plant should be balanced with its production scale, the available land resources, and the ease of transporting of feedstock and products (Shepherd, 2009).
2) Investment in biomass pyrolysis: in the pyrolysis technique, the biomass needs to be separated (i.e. urban municipal waste), compacted (i.e. energy crops such as hybrid popular), along with assembling and hauling, which altogether would materially affect

the cutting fee, tipping fee and the storage cost (Joseph and Lehmann, 2009). Moreover, the feasibility to use biomass energy generated from biomass pyrolysis may depend upon how much the local people rely on it. For instance, people who live in rural Africa and the countryside of Asia and Latin America used to utilize biomass energy so the replacement of wood burning by pyrolysis energy can be an alternative as clean energy to mitigate carbon emissions for those regions (Joseph and Lehmann, 2009).

3) Contaminants: the biochar produced from nutrient-rich materials such as animal-manures and municipal waste could contain toxic heavy metals and polycyclic aromatic hydrocarbons (PAH) (Shackley et al., 2010, Kuppusamy et al., 2016)
4) Damage to soil: large-scale biochar application could cause soil compaction (Shackley et al., 2010). Firstly, biochar is easily transported to the deeper soil due to its lighter density, which carries the pollutant particles absorbed into the biochar surface to the subsurface profile, so the concentration of pollutants in the underground water would increase. Secondly, biochar-amended soils could become a source of contaminants associated with a reduced degradation due to their strong sorption.
5) Binding and reducing the efficacy of agrochemicals: some articles have acknowledged that the crop yield is not as high as expected when biochar is present unless N/P fertilizers are used (Kuppusamy et al., 2016), and also the sorption of biochar might hinder the bioactivation of insecticides and pesticides. Plus, some biochar with C/N ratios above 25-30 may cause N immobilization in soils (Sullivan and Miller, 2001) and nearly 50% of P in biochar is less plant available resulting from its HCI-extractable form, which is more apparent in higher temperature pyrolysis biochar (Bridle and Pritchard, 2004, Chan and Xu, 2009).
6) Unbalanced uptake of plant nutrients: organic and inorganic chemicals retained in biochar-introduced soil may be maintained at an uneven level, although preventing the leaching of nutrients is essential (Joseph and Lehmann, 2009). Particularly, retaining a large amount of cation is difficult in freshly produced biochar (Chan and Xu, 2009). Furthermore, crop productivity might be impacted adversely because the rising pH destroys the micronutrients environment, which would ultimately threaten food security (Chan and Xu, 2009).
7) Technological constraints in the biochar application. Practical deployment of biochar on a large scale contains multiple considerations. For instance, chemical and physical characteristics of fields, the time to apply, such as whether it should align with the farming schedule, the availability of mechanical equipment, how to conduct the pre-

treatment; such as grinding and adjusting soil water content, working hours during application, and how would working hours affect the application rate in every single project (Shackley et al., 2010).

8) Some industries lack the carbon emissions trading standard. According to the guidelines of Clean Development Mechanism, the trading C sequestration programme in arable land is not allowable, which constrains the popularity of biochar practice in agriculture (Lehmann et al., 2006).

2.7 Policy and Economic Implications of Carbon Capture

Since parties comprising the IPCC have signed its Fifth Assessment Report in 2014, which aimed to limit the increase of global warming to 1.5°C above pre-industrial levels, each member country has regulated the upscaling and ambitious adaption and mitigation pathways to achieve “net-zero” carbon emissions. In the UK, the Government and Parliament adopted the recommendations of the Committee on Climate Change that UK will achieve net-zero GHG emissions by 2050 (Stark et al., 2019). Correspondingly, Scotland has set a net-zero GHG target by 2045 and Wales has proposed to reduce the GHG emissions by 95% by 2050 against 1990 levels with an ambitious target of net-zero GHG by this date (Stark et al., 2019). Japan has decided to reach a “carbon neutral society” as early as possible during the latter half of this century; the USA aims to accomplish net-zero by 2050; the Finnish Government has announced its target to reach carbon-neutrality by 2035 (Stark et al., 2019). However, large uncertainties exist regarding carbon capture, mainly related to the cost, feasibility, sustainability and side-effects on the environment (Masson-Delmotte et al., 2018), hence any deployment of carbon removal technology would carry associated risks. No matter which options are pursed to achieve carbon emissions reduction, or the enabling conditions supporting the implementation of carbon management plans, all meet similar knowledge gaps and barriers (Figure 2. 7). For example, in land ecosystem transition, how to allocate land demands and availability to fulfil carbon mitigation action is always an overarching issue; meanwhile, the non-technical reasons such as vested interests will, importantly, determine the participatory attitude of the cooperating partners (Shepherd, 2009). In the UK, although the net carbon account emissions dropped slightly under the carbon budget programme (2013-2017), e.g. a 2% fall in 2018, there is still a considerable shortage for some sectors to meet their individual goals. For instance, the contributions from agriculture, land use and forestry across the UK on decreasing carbon budgets is less encouraging as they have not met the indicators stated in the budget programme (Stark et al., 2019). As for green buildings improvement such as solid walls insulation, heat pump installation and low-carbon heat, the changes made for alleviating carbon emissions is still not satisfied. Despite good progress accomplished in the power and waste fields regarding the carbon budget, a lacking of firm government policy is the dominant factor to prevent the achievement of another budget aims on schedule (Stark et al., 2019).

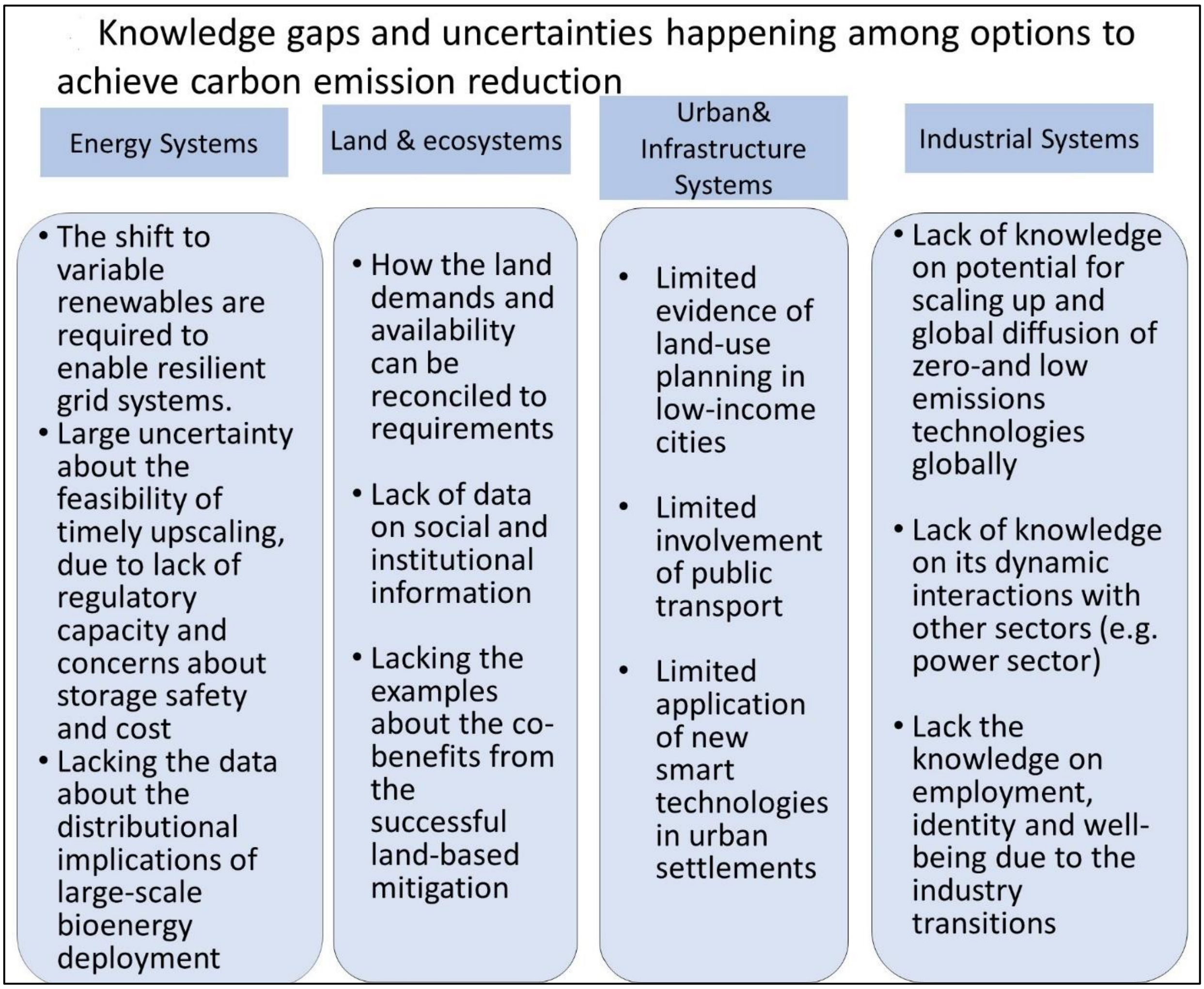

Figure 2. 6 Knowledge gaps and uncertainties in terms of carbon emission reduction occurring in options (Masson-Delmotte et al., 2018) (Adapted by Jiaqian Wang).

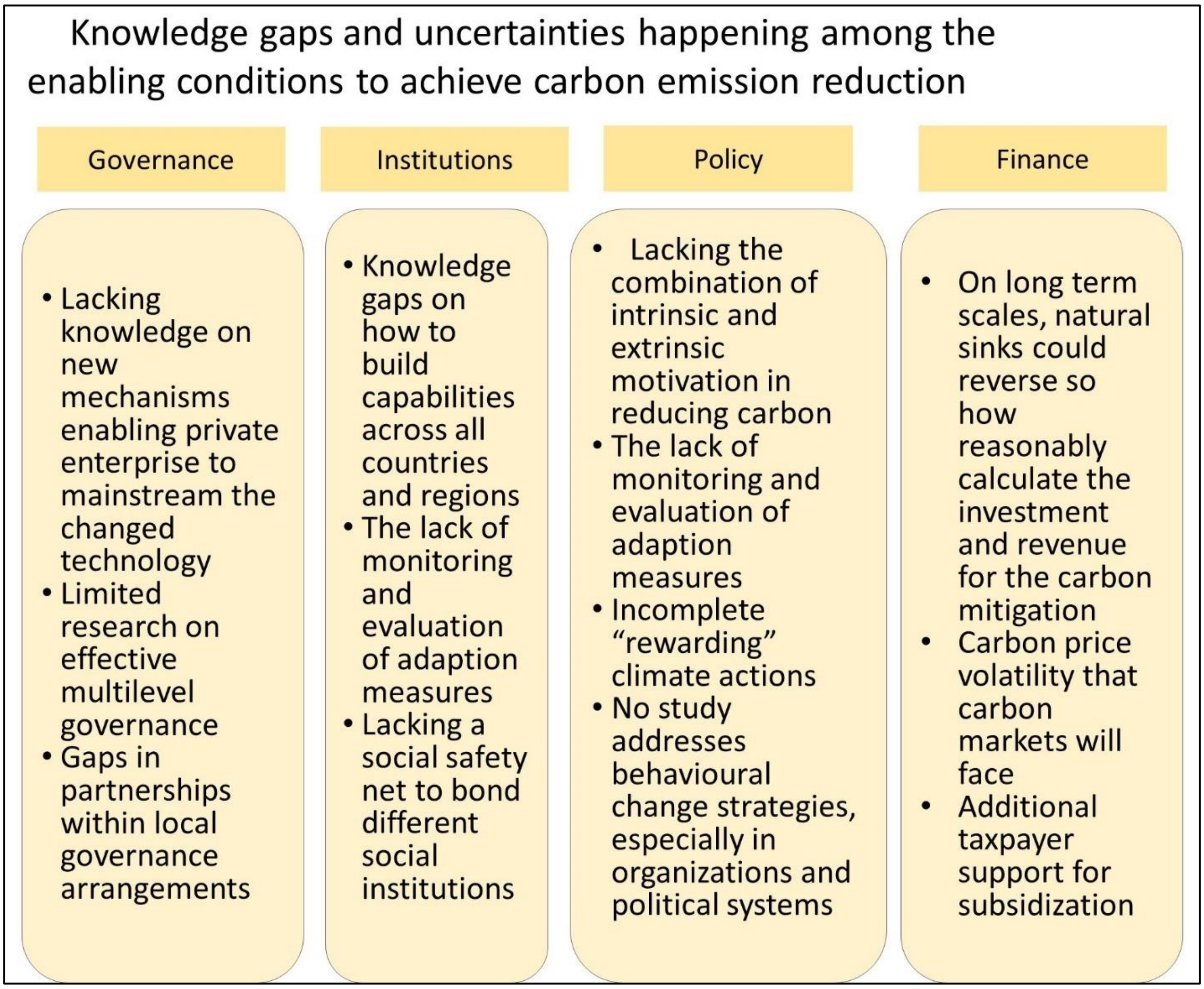

Figure 2. 7 Knowledge gaps and uncertainties in terms of carbon emission reduction occurring in enabling conditions (Masson-Delmotte et al., 2018) (Adapted by Jiaqian Wang).

2.8 The relevance to this research

Numerous projects have proved that the increase in atmospheric CO_2 concentrations has a detrimental effect on the planet, and many announcements have urged people to play a part in mitigating climate change. The interaction of carbon with other chemical compounds and creatures among different ecosystems and within an individual ecosystem is complicated, so the situation of carbon stock and carbon sequestration potential would vary with region. Although some institutions will introduce a carbon management plan to reduce carbon from the carbon production and emission perspective, few of them have considered quantifying the carbon stock baseline from their estate to assess the potential magnitude of carbon sequestration and capture. In the UK, there is a lack of studies which analyse how institutional management of estates (such as by universities, companies, and local councils, etc.) affects the carbon stock from the point of view of land usage or land conversion. Furthermore, this absence hinders the ability to predict the quantity of carbon which could be sequestered via the change of land cover or estate management. Newcastle University provides an interesting case study in this context, because it has a campus in the heart of the city of Newcastle in north-eastern England, involving urban brownfield site reclamation, and also takes charge of urban sports grounds, and two farms with crop fields and woodlands. Its estate thus provides a very diverse, but easily accessible and readily available research facility to categorize the variety of carbon storage opportunities present in each separate environment.

In addition to establishing the carbon storage baseline, the feasibility to capture carbon from the atmosphere or store more carbon in the available terrestrial land via changes in land management should be taken into account. Also, biochar technology is increasingly considered regarding waste utilization and carbon removal. As farms run by Newcastle University produce a great amount of crop and manure residues every year, there will be an opportunity to investigate the effectiveness of operating an internal institution waste recycling plant to produce biochar, and then deploying biochar on the campus or over the arable areas as well to improve the soil carbon sequestration. Biochar application has resulted in various arguments in the literature about the investment, feasibility, and revenue. Therefore, according to our experimental outcome, how these arguments would change in the biochar market in England will be examined. Considering the Net-Zero Carbon aim of Newcastle University, we will suggest to carry out the biochar scheme as well as land conversions with different scenarios across the institutional lands. Then, in the following chapters we will discuss the opinions regarding to our proposals, of directors and leaders of the university who are responsible for city campus estate, farm, and sustainability management.

3. Chapter Three. Carbon sequestration potential and analysis of farming methods on public farms

3.1. Introduction

The Paris Agreement has built consensus amongst 197 state parties to limit the increase of global average temperature to 1.5 ℃ above pre-industrial levels (UNFCCC, 2016). Soil carbon management has a vital role to play in achieving this goal, exemplified by the “4 per 1000” initiative (Lord and Sakrabani, 2019, Minasny et al., 2017). Government aims to mitigate climate change would be unachievable if contributions from individual organizations were absent (Knuth et al., 2007). Many institutions, including universities, also recognize the need to address the climate emergency (Knuth et al., 2007, Lewis and Patton, 2010, Mazhar et al., 2014, Robinson et al., 2018). When the Climate Neutral Network (CN Network, 2009) was launched by the United Nations Environment Programme in 2008, six universities from the USA, UK, Spain and China committed to building low-carbon campuses (Shin, 2009). By December 2013, 669 academic institutions became signatories of The American College and University Presidents' Climate Commitment (ACUPPC) which aims to reduce 80% of greenhouse gases (GHG) emissions by the middle of 21st century (Delaney, 2010, Peterson, 2013). This is important, because approximately 2 % of the GHG in the USA are produced from colleges and universities (Shin, 2009, Sinha et al., 2010). In 2018/19, a total of 161 universities in the UK emitted nearly 11 million metric tonnes of CO_2, constituting 3% of UK emissions (Mitchell-Larson et al., 2021). In the UK, many English universities have launched ambitious carbon management plans, as required by the Higher Education Funding Council for England, with similar plans in Scottish and Welsh universities (Lewis and Patton, 2010). These commitments show how academic institutions globally can voluntarily contribute to national and multi-national climate change mitigation plans and set an example for other institutions. While the important role of universities in national carbon emission reduction plans has been acknowledged widely (Mazhar et al., 2014, Mitchell-Larson et al., 2021, Robinson et al., 2018), some universities may not achieve their ambitious carbon reduction goals (Warner, 2016).

As more institutions adopt ambitious net-zero or neutral targets for their future carbon emission, plausible carbon off-setting strategies become increasingly important. In their latest briefing at the 26th United Nations Climate Change Conference (Mitchell-Larson et al., 2021), climate change experts have emphasized that higher education institutions should carry out nature-based carbon removal such as growing trees and restoring forests at scale. However, a review of sixteen university carbon management schemes available online showed that none considered terrestrial carbon, including carbon stored in soils and by plants, in a quantitative way (Table A. 1 and related discussion **in Appendix A1**). Nevertheless, many academic institutions have substantial land holdings. For example, Newcastle University occupies an urban campus of around 25 hectares in north-eastern England, but more significant in terms of its institutional land management are two research farms, Cockle Park Farm and Nafferton Farm, with a total land area of 805 hectares. As Newcastle University is working towards net-zero carbon dioxide emissions by 2030 (Newcastle University, 2021) , it seems pertinent to consider management opportunities for the entire estate to capture and store atmospheric CO_2, setting an example for institutions globally with significant land holdings. For example, Oxford and Cambridge Universities are amongst the largest landowners (23,151 and 18,433 ha, respectively) in the UK (Barbiroglio, 2018). The ten largest college campuses in USA cover above 45,982 hectares (Egan, 2019). Many other government and non-government organizations and private sector institutions also own significant amounts of land. For instance, amongst the private water companies in England, United Utilities has the largest land holding of around 57,061 hectares (Shrubsole, 2016). Local authorities own approximately 4% of land in England (Shrubsole, 2020). In Scotland, approximately 32,780 hectares of land are owned by 32 councils (Picken and Nicolson, 2019). Local authorities in Wales own land used for farming purposes with just over 16,441 hectares (Welsh Ministers, 2018). There are 125,857 hectares of golf course in Great Britain, which is similar to the whole public park area (125,048 hectares), and the majority of these golf courses are owned by local authorities (Shrubsole, 2020). A recent questionnaire survey of twenty-seven local authorities from across the UK revealed that 81% of the councils had declared a climate change emergency and 70% had committed to additional tree planting, but only one council had related its tree planting target to carbon emissions across the authority (Ross, 2020).

Optimized land management has a significant potential for greater carbon sequestration (Kaplan et al., 2012, Rees et al., 2018, Wang et al., 2019, Wiesmeier et al., 2019). The “4 per 1000” initiative for example seeks to increase soil organic carbon globally by the annual rate of 0.4% to compensate the GHG emissions resulting from human activities (Lord and Sakrabani, 2019, Minasny et al., 2017). However, achieving the “4 per 1000” goal is a formidable challenge in temperate regions, as has been exemplified with agricultural field experiments in the south-eastern UK (Poulton et al., 2018), and this applies even more so in the northern UK, where soil C content is already higher than in the south (Bradley et al., 2005, Feeney et al., 2021). Nonetheless, the soil organic carbon pool has experienced substantial losses under agricultural management, but could reach an equilibrium in other ecosystems such as forests or prairies (Jarecki and Lal, 2003). Approximately 40% of radiative gases in the atmosphere result from agricultural activities and conversion of land use (Ward and Mahowald, 2014). Globally, poor management of land change and cultivation practices could cause more carbon emissions than the combustion of fossil fuels (Lal, 2003). Consequently, it is imperative that soil, as part of land management overall, is considered as part of institutional carbon management plans, especially for institutions with significant estates.

The aim of this chapter was to develop and demonstrate a methodological framework for quantifying and managing terrestrial carbon on institutional estates. The objectives of the framework are i) to establish the current carbon stocks of estates as a database and future reference point, ii) to obtain from the integration of these data with land use records a quantitative understanding of how management affects terrestrial carbon stocks, and iii) to derive from this analysis realistic and locally appropriate strategies for achieving institutional carbon reduction goals by changes in land management.

3.2. Methodology

3.2.1 Methodological framework

The methodological framework of this study is illustrated in Figure 3. 1. First, the current soil and tree carbon stock on the institutional estate was surveyed to establish a baseline for future reference and a dataset for the analysis of land use effects. Next, these field data were integrated with the institutional and publicly available land use records to derive quantitative understanding of land management effects on terrestrial carbon in the institutional estate. Finally, the future terrestrial carbon stores were predicted as a function of future land management scenarios and quantitatively related to the institutional carbon emissions and reduction targets. The annual carbon emissions (CO_2 equivalents-C) for Newcastle University in the academic year 2019/20 were obtained from its carbon management plan (Newcastle University, 2021).

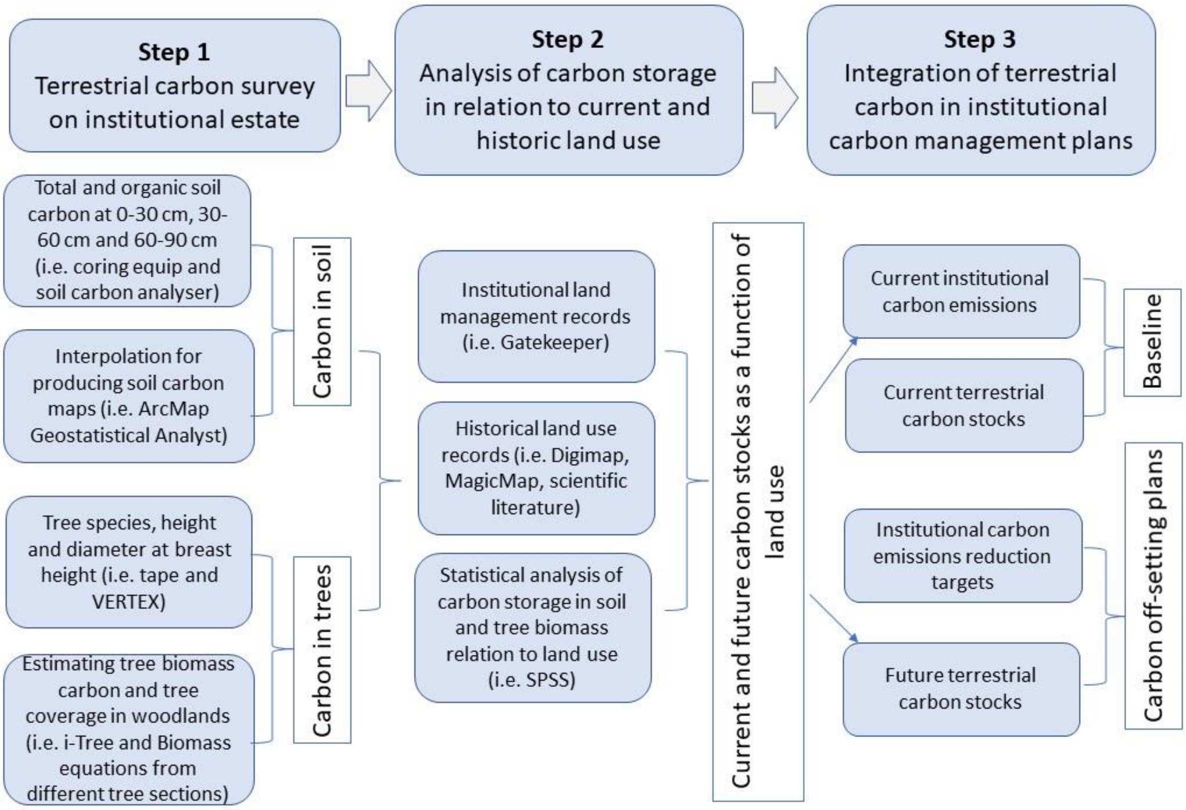

Figure 3. 1 Schematic diagram of methods used to inform management of estate carbon stocks.

3.2.2 Soil and tree carbon survey

The soil collection work was conducted in April 2018 (arable land and permanent grassland, Cockle Park Farm), March 2019 (arable land and permanent grassland, Nafferton Farm), October 2019 (woodlands, Cockle Park Farm), and February 2020 (woodlands, Nafferton Farm). The sampling locations were evenly distributed over agricultural land and woodlands, and every soil type were covered. Overall, 102 points were sampled across 2 farms: 55 points were sampled at Cockle Park Farm (39 plots in agricultural land versus 16 plots in woodlands), which resulted in 163 soil samples (approx. 350 g each sample). At some locations soils from 60-90 cm could not be obtained due to obstacles encountered when coring (2 sampling points with red circle in Figure A3). Similarly, there were 139 soil samples from three soil depths at 47 sampling points at Nafferton Farm (31 plots in agricultural land versus 16 plots in woodlands). Soil was sampled at three depth increments (0-30 cm, 30-60 cm, and 60-90 cm) with tractor mounted coring equipment in crop fields and using a hand auger in woodlands. The coring equipment or hand auger was drilled to a depth of up to 1 m underground. Below 1 m soil layer, rocks were found in most areas and mud or sands were found in a few places. After coring the tube with the drillings, the soil core was placed horizontally on the ground. A tape measure was used to divide the soil cores into 30 cm intervals where the first segment (0-30 cm depth) started from the top of the tubes. Each sample was then placed and sealed in an individual zip-top plastic bag. After sampling, the soils were moved back to the laboratory and stored at 4 ºC in a cold room. Large stones, roots and other plant debris were removed before oven-drying the soils for about 48 hours at 105 ℃ to a constant weight, while recording the loss of weight as the water content of the soils. Afterwards, samples were passed through a 4.75 mm sieve because soil macroaggregates below this size drive the long-term carbon sequestration with high resistance to erosion (Blanco-Canqui et al., 2017). The dried and sieved samples were milled to a fine powder for 2 minutes (Laboratory Disc Mill, TEMA Machinery Ltd, UK). For comparison, a few samples were ground by hand with a mortar and pestle, and this yielded similar results. The samples were analysed for carbon as percent mass by a dry oxidative combustion procedure at up to 1000 ℃ using the LECO RC 612 analyser (LECO Corporation (2018); Saint Joseph, Michigan USA). Ex-situ bulk dry soil density was calculated in the laboratory by considering the sieved, dry soil mass obtained on average for the core volume from each soil depth layer. The carbon density was calculated from the carbon content and the ex-situ bulk dry soil density. Additionally, soil was analysed for pH as well. More detail about the soil carbon calculations, pH analysis, and the division of soil types is provided as **Appendix A2.** Carbon distribution maps were made using ArcMap (version 10.6.1) with geostatistical analysis extension, as explained in **Appendix A2**. To assess the carbon stored in tree biomass, the parameters such as tree diameter at breast height, height, and species were obtained for a total of 117 trees within 6 surveying plots at Cockle Park Farm, and another 30 trees at Nafferton Farm. The software package i-Tree (2020) from the United States Department of Agriculture Forest Service, which includes the tools i-Tree Eco and i-Tree Canopy, then enabled quantification of carbon in individual trees, land tree coverage, and ultimately the woodland biomass. For comparison with i-Tree Eco, the Woodland Carbon Code: Carbon Assessment Protocol of the Forestry Commission of England (Jenkins et al., 2018) was also used to estimate the carbon storage of trees. The carbon stored by trees in the woodlands was calculated by multiplying the whole area of tree cover obtained from i-Tree Canopy, and the carbon storage of the trial plots obtained from i-Tree Eco. Also, individual trees and small groups of trees grew along field edges and in some fields at the two farms. The crown area of these trees was estimated on satellite images on Google Earth, and multiplied by the mean carbon storage of the woodland trees, to calculate their carbon stocks. More detail about the tree biomass carbon surveying methods is provided as **Appendix A2**. Not all of the soil sampling locations in the woodland were within the plots where biomass was measured. The selected biomass measurement sites only occupied a portion of the woodland area (Figure 3.3& Figure 3.4), whereas the soil sampling locations were distributed across the entire woodland area. Additionally, in December 2020, we interviewed the farm director of Newcastle University to understand the current farm management practices and management constraints on options for arable land conversion. The interview text is provided in **Appendix D**.

3.2.3 Statistical data analysis of land management effects

Institutional crop rotation records from Gatekeeper (2020), a software package for farm management, a map illustrating land use at Cockle Park farm in approximately 1900 (Shiel, 2000), and other historic maps (Digimap, 2020, MAGIC, 2020) were used to study relationships between land management and terrestrial carbon. At Nafferton Farm, management between 2002 and 2017 divided the farm into a conventional and organic system. At two farms agricultural land managed for at least five years as permanent grassland could be distinguished from the arable land. The woodland could be classified as either coniferous or broadleaved by polygon areas according to maps on MAGIC. Woodland at Cockle Park Farm could be distinguished according to the time of establishment, which was estimated from historic maps. The responses between means of continuous variables (total carbon, organic carbon and pH) to variations of independent factors such as land management were tested by the univariate analysis where it applied the Tukey's HSD in SPSS (IBM crop, 26.0), and differences were considered significant for a p-value ≤ 0.05. A formula was then developed from the field data using the average amount of soil and tree carbon per m^2 of surface area for each land use type to predict the total carbon stocks at the two farms as a function of land management. More details about the field data categorization and evaluation are provided as **Appendix A2**.

3.3. Results

3.3.1 Terrestrial carbon stocks in agricultural land and woodlands at the two university-run farms

The total terrestrial carbon stock of the agricultural land (top 90 cm of soil) and woodlands (trees and top 90 cm of soil) at the two university-run farms amounted to 103,620 tonnes (Table 3. 1). This carbon stock was equivalent to sixteen times the carbon emissions of Newcastle University in 2019/20 (6,406 tonnes of CO_2 equivalents-C) (Newcastle University, 2021). Eighty-nine percent of this carbon stock was in the 90 cm soil layer of arable land and permanent grasslands, with over half of that carbon located in the top 30 cm soil layer (Figure

3. 2). Six percent of the total terrestrial carbon stock was in the top 90 cm of woodland soil. Woodland trees accounted for four percent, and 'hedgerow trees' for one percent of the total terrestrial carbon stock.

3.3.2 Factors influencing total carbon, organic carbon density and pH in soil

Soil total and organic carbon densities expressed in Kg·m^{-3} and soil pH across three soil depths at the two farms, overall, and differentiated according to land use, are summarized in Table 3. 2. TOC accounted for ⩾90% of the reported TC. Since soil inorganic C was comparatively small and could reflect geological sources (such as limestone fragments in the parent glacial till), it was not interpreted separately. Generally, total carbon (TC) and total organic carbon (TOC) decreased with soil depth on both farms (One-way ANOVA, $p<0.001$, Table A. 2 **in Appendix A2**), but the differences between the 30-60 cm and 60-90 cm layers were not statistically significant (Tukey's HSD in univariate analysis, $p>0.05$, Table A. 3 **in Appendix A2**). Additionally, significant interacted influence caused by soil depth and the classification of agricultural land on the carbon value existed at both farms (One-way ANOVA, $p<0.01$, Table A. 2 **in Appendix A2**). Over the woodland, at least one soil sample collected in each biomass surveying plot, but an insufficient number for deriving soil carbon for each type of tree coverage (e.g. Figure A. 7). However, sufficient sampling points was available to compare coniferous and broadleaved woodlands. Consequently, there were no statistically significant differences in soil carbon densities when comparing soils of broadleaved and coniferous woodland at either farm (One-way ANOVA, $p>0.05$, Table A. 2 **in Appendix A2**).

		Unit	CPF	NF	Rural estate
Carbon Storage	Soil carbon storage top 90 cm of permanent grassland	$Kg \cdot m^{-2}$	12.14	17.13	14.67
	Soil carbon storage top 90 cm of arable land	$Kg \cdot m^{-2}$	10.30	12.16	11.52
	Soil carbon storage top 90 cm of coniferous woodlands	$Kg \cdot m^{-2}$	15.30	16.64	15.76
	Soil carbon storage top 90 cm of broadleaved woodlands	$Kg \cdot m^{-2}$	13.25	16.34	14.29
	Biomass carbon storage coniferous woodlands	$Kg \cdot m^{-2}$	12.68	12.60	12.65
	Biomass carbon storage broadleaved woodlands	$Kg \cdot m^{-2}$	10.65	n.a †	10.65
	Biomass carbon stock per hedgerow tree	Kg	395.00	395.00	395.00
Field Area	Land area permanent grassland	hectares	60.2	61.9	122.1
	Land area arable land	hectares	221.8	423.1	644.9
	Land area coniferous woodlands	hectares	18.7	9.8	28.5
	Land area broadleaved woodlands	hectares	6.3	3.2	9.5
	Number of hedgerow trees	trees	1146	1260	2406
Carbon stock	Soil carbon in top 90 cm of permanent grassland	tonnes	7306	10601	17907
	Soil carbon in top 90 cm of arable land	tonnes	22854	51440	74294
	Soil carbon in top 90 cm of coniferous woodlands	tonnes	2865	1632	4496
	Soil carbon in top 90 cm of broadleaved woodlands	tonnes	831	522	1353
	Biomass carbon in coniferous woodlands	tonnes	2375	1235	3611
	Biomass carbon in broadleaved woodlands	tonnes	668	340	1008
	Biomass carbon in hedgerow trees	tonnes	452.7	497.7	950
	Total terrestrial carbon	**tonnes**	**37352**	**66268**	**103620**

Table 3. 1. Carbon storage ($Kg \cdot m^{-2}$), field areas and carbon stock of different ecosystem components at Cockle Park Farm (CPF) and Nafferton Farm (NF).

†: no tree parameters measurement at Nafferton broadleaved woodlands due to the restrictions of ground situations.

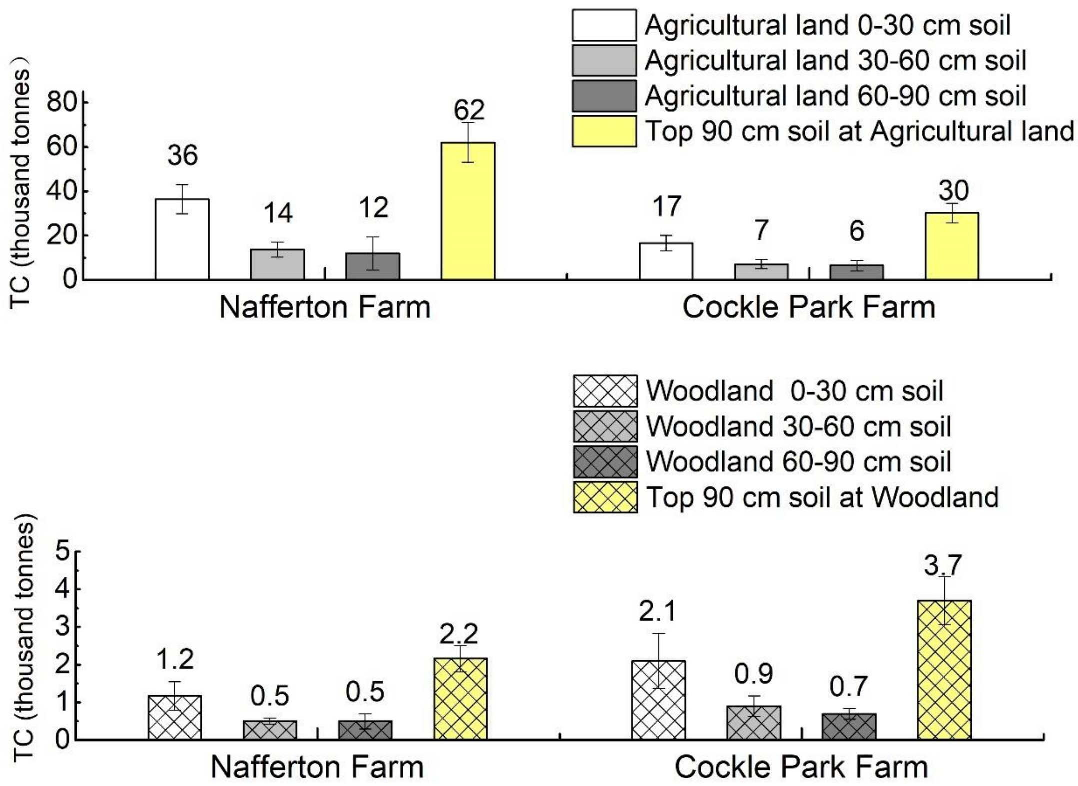

Figure 3. 2. Total carbon stocks (thousand tonnes, Mean ± SD) at three soil depths and for the top 90 cm at the two university-owned farms.
SD: Standard Deviation.

At Cockle Park Farm, the topsoil of permanent grassland could store more carbon than that of arable land (One-way ANOVA, TC: p=0.004; TOC: p=0.002, Table A. 5 **in Appendix A2**). At Nafferton Farm, there was no statistically significant difference between organic and conventional management (One-way ANOVA, p>0.05, Table A. 5 **in Appendix A2**). At both Nafferton Farm and Cockle Park Farm, higher TC and TOC density was observed in woodland compared to agricultural land soil for all soil depths (One-way ANOVA, p<0.05, Table A. 6 **in Appendix A2**). Correlation analysis revealed that the variance for TC and TOC over the woodlands mainly resulted from soil depth rather than farm location, or the combined effect of these two variables (Tukey's HSD in univariate analysis, Table A. 4 **in Appendix A2**). When comparing soil from a woodland at Cockle Park Farm established after 1960 with soil from a woodland established since 1860, the differences in TC and TOC were not statistically significant for any of the soil layers, although a higher mean soil carbon density was found in the older woodland (Table 3. 2). This may reflect that soil carbon increases only incrementally after 40 years of land management as woodland. Note that the woodland age is not necessarily equivalent to tree age because of replanting, and the average tree age estimated from DBH was comparable across the woodlands.

Soil pH increased with the soil depth at both farms (One-way ANOVA, $p<0.001$, Table 3. 2 & Table A. 2). Soil pH in crop fields was generally higher than in woodlands, although a significant correlation was only found at Nafferton Farm (One-way ANOVA, $p<0.001$, Table A. 2 **in Appendix A2**). In each soil profile, pH significantly related to soil carbon storage on both farms with negative correlation coefficients, excluding TC at 60-90 cm soil (correlate analysis, $p<0.05$, Table A. 7 **in Appendix A2**).

Site	Land use	Number of samples	Soil carbon density (Kg·m^{-3})												pH					
			0-30 cm				30-60 cm				60-90 cm				0-30 cm		30-60 cm		60-90 cm	
			TC		TOC		TC		TOC		TC		TOC							
			Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
CPF	Permanent grassland	9	24.64	7.16	23.44	6.78	8.43	2.27	7.90	2.10	7.41	1.18	6.94	1.14	6.59	0.28	7.20	0.61	7.25	0.69
	Arable land	30	18.26	4.82	17.21	4.58	8.41	3.02	7.76	2.67	7.67	3.64	6.61	2.98	6.77	0.41	7.34	0.44	7.54	0.63
	Coniferous woodland	10	29.49	12.68	28.21	12.43	12.47	4.78	11.71	4.62	9.03	2.30	8.46	2.20	5.12	0.85	5.53	0.70	6.03	0.68
	Broadleaved woodland	6	23.75	7.71	22.57	7.47	10.68	2.71	10.07	2.59	9.74	2.64	9.02	2.49	4.70	0.24	5.41	0.44	6.03	0.68
	Long established woodland	10	29.18	9.77	27.87	9.51	12.79	4.91	12.07	4.72	9.51	2.72	8.84	2.60	4.74	0.46	5.33	0.59	5.95	0.81
	Recently established woodland	6	24.27	13.55	23.14	13.35	10.16	1.53	9.46	1.37	8.94	1.82	8.38	1.68	5.33	0.93	5.75	0.57	6.15	0.30
NF	Permanent grassland	2	38.41	9.88	32.30	3.74	11.49	5.52	9.92	4.30	7.19	n.a †	6.15	n.a †	5.66	0.34	6.85	1.15	7.84	n.a. †
	Arable land	29	23.08	4.96	20.96	4.43	9.07	2.54	7.83	2.50	8.38	5.25	6.21	2.24	6.66	0.37	7.30	0.43	7.58	0.45
	Coniferous woodland	5	30.37	12.28	28.69	11.96	11.83	2.07	10.86	1.69	13.27	6.68	12.27	6.50	5.60	1.20	6.69	0.60	6.51	1.04
	Broadleaved woodland	11	28.39	11.44	26.54	10.79	15.35	4.59	13.90	4.61	10.71	2.63	9.74	2.06	5.49	0.84	6.36	1.04	6.88	0.56
	Conventional	16	25.30	7.10	22.55	5.21	9.71	3.35	8.46	2.77	7.94	4.93	6.09	2.75	6.50	0.45	7.09	0.49	7.36	0.42
	Organic	15	22.75	5.48	20.77	5.18	8.72	1.81	7.44	2.36	8.82	5.60	6.36	1.34	6.69	0.41	7.47	0.40	7.88	0.28

Table 3. 2 Soil Total Carbon density (TC, Kg·m^{-3}; Mean ± SD), Total Organic Carbon density (TOC, Kg·m^{-3}; Mean ± SD) and Soil pH (Mean ± SD) at Cockle Park Farm (CPF) and Nafferton Farm (NF). SD: Standard Deviation.

†: Only one 60-90 cm soil core was sampled at Nafferton Farm permanent grassland because the other one was too compacted to collect.

3.3.3 Geospatial distribution of total and organic soil carbon

Spatial distribution maps of TC and TOC densities on the two farms illustrate how the woodlands strongly influenced the overall soil carbon distribution at both farms, and the highest concentration of TC and TOC in all three soil layers was generally measured in the woodlands of both farms (Figure 3.3 & Figure 3.4). The agreement between interpolated and measured carbon values is shown in Figure A. 1-6 **in Appendix A2**. At Cockle Park Farm, the density of TC ($Kg \cdot m^{-3}$) and TOC ($Kg \cdot m^{-3}$) showed a similar distribution in the 0-30 cm and 30-60 cm soil layers, being greater in the centre along an east-westerly direction as compared to other places (Figure 3.3). In the 60-90 cm soil layer, the predicted distribution map of carbon showed higher carbon densities in the extreme western parts, whereas lower carbon contents were measured in the centre along a north-south direction at Cockle Park. At Nafferton Farm, a small part on the western fields showed a high carbon density comparable to the woodlands. In the 30-60 cm soil layer, soil carbon density was again greatest for the woodland sites. As for the 60-90 cm soil layer in Nafferton Farm, TC density, instead of TOC, was the highest in the central part of the farm rather than in the woodlands. This discrepancy may be caused by the existence of rocks below the soil sampling depth or a more enabling condition for Ca^{2+} precipitation. Meanwhile, in this area of the farm, coal outcrops beneath the soil, and it is likely that fragments of coal have contributed to the determined TC within the deepest samples.

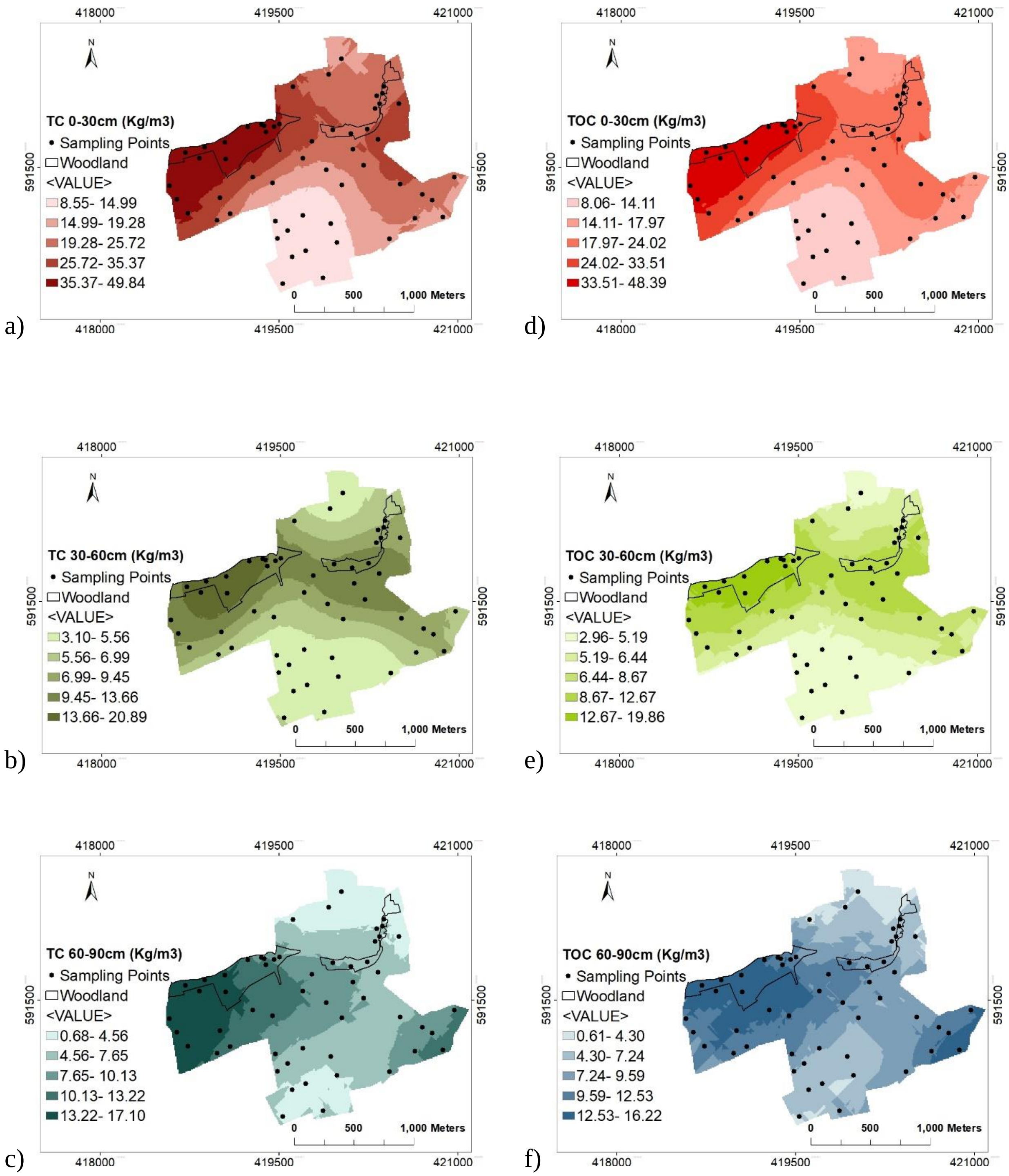

Figure 3. 3. Interpolated maps using Ordinary Kriging for the density distribution of soil total carbon (TC) and total organic carbon (TOC) at Cockle Park Farm.

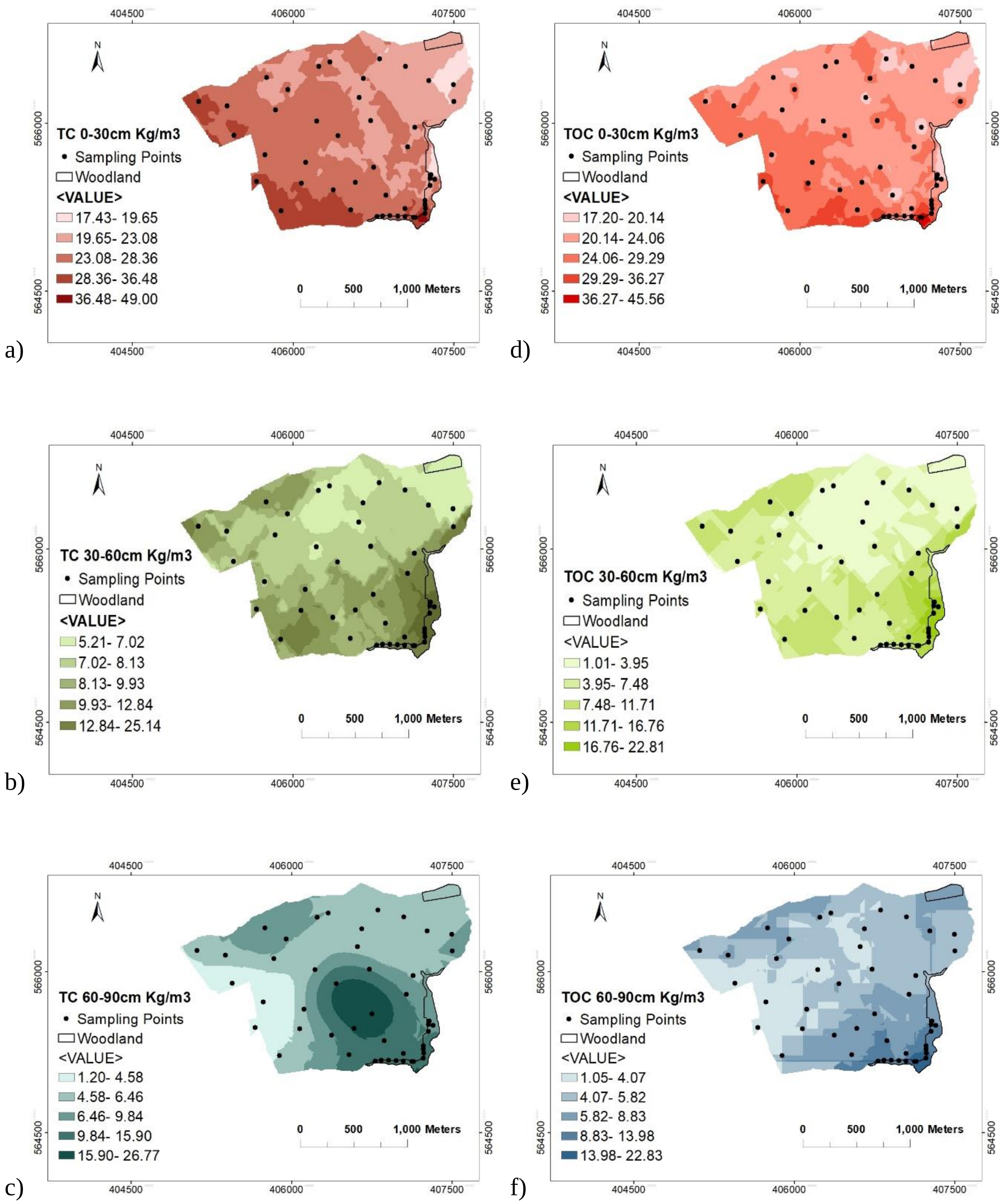

Figure 3. 4. Interpolated maps using Ordinary Kriging for the density distribution of soil total carbon (TC) and total organic carbon (TOC) at Nafferton Farm.

3.3.4 Factors influencing carbon stored in woodland trees at the two university-owned farms

Table 3. 3 presents the tree data collected at Cockle Park Farm and Nafferton Farm which include tree species, survey area, DBH, tree height, and carbon storage of various tree types processed in i-Tree Eco, and also using the biomass equations of the Woodland Carbon Code (Jenkins et al., 2018). Additionally, it shows the calculated total carbon storage in the woodlands at the two farms. The regression correlation of tree carbon stocks between i-Tree Eco and biomass equations was: y = 0.87x + 1.39 (y: trees carbon stores from biomass equations; x: trees carbon stocks from i-Tree Eco; R^2=0.83), which showed comparable carbon stock results from two approaches. Across the whole i-Tree dataset, stands of Norway Spruce (15.73 $Kg{\cdot}m^{-2}$) and Sitka Spruce (15.29 $Kg{\cdot}m^{-2}$) at Cockle Park Farm exhibited the highest mean C biomass storage, and European Larch stands contributed the lowest C biomass storage (7.03 $Kg{\cdot}m^{-2}$). According to i-Tree Eco, the average carbon storage on the woodlands at Nafferton Farm was 12.60 $Kg{\cdot}m^{-2}$, slightly higher than the 11.67 $Kg{\cdot}m^{-2}$ at Cockle Park Farm. The tree canopy coverage areas of woodlands at Cockle Park Farm and Nafferton Farm according to i-Tree Canopy were 25 ha and 13 ha, respectively (Table 3. 3).

Site	Species	Number of Trees	Fieldwork Plot Area (m^2)	Average Height (m)	Average DBH (cm)	i-Tree Total Carbon Storage (tonnes)	Biomass equations Total Carbon Storage (tonnes)	i-Tree Average Carbon Storage ($Kg \cdot m^{-2}$)
CPF	European Larch	20	583	20.74	34.93	4.1	6.30	7.03
	Sycamore	20	585	21.45	34.11	7.5	8.61	12.82
	English Oak	17	1000	24.36	36.77	9	11.23	9
	Sitka Spruce	20	340	20.01	33.32	5.2	4.95	15.29
	Norway Spruce	20	89	15.96	17.98	1.4	1.09	15.73
	Mix (Sycamore& English Oak)	20	780	18.35	33.31	7.9	8.67	10.13
	Sum-entire study area	117	3377	20.15	31.74	35.1	40.85	11.67
NF	Mix (Sitka Spruce& Norway Spruce)	30	1032	18.04	41.08	13	10.89	12.60
		Area (hectares)				Carbon Storage (tonnes)		
CPF	Trees in the woodlands	25				3,043		
NF		13				1,576		
CPF	‘Hedgerow’ trees	3.89				452.7		
NF		3.95				497.7		

Table 3. 3 Carbon storage ($Kg \cdot m^{-2}$) of individual tree species in the fieldwork plots in the woodlands, the overall estimated carbon stock (tonnes) of trees in the entire woodlands and the estimated carbon stock of ‘hedgerow’ trees at Cockle Park Farm (CPF) and Nafferton Farm (NF).

3.3.5 Scenarios for carbon-offsetting by terrestrial carbon augmentation in the institutional estate via changes in land management, and related opinions of the farm manager

While most of the terrestrial carbon is currently stored in the agricultural of the university farms, the woodlands stored significantly more carbon per square meter than the fields (Table 3. 1). The difference between the two land use types (the subtraction of the mean for agricultural land from the mean for the woodland), was 14.72 $Kg \cdot m^{-2}$ at Cockle Park Farm and 14.45 $Kg \cdot m^{-2}$ at Nafferton Farm (Table 3. 1). Using the insights gained from the current carbon stock surveys and analysis, the total carbon stock for alternative land use scenarios at the two farms can be estimated. Totally, four scenarios were developed for offsetting a portion of Newcastle University's carbon emissions (CO_2 equivalents-C) by changes in land management on its estate (Table A. 8-11).

Under scenario 1, if the entire university farm sites were converted to coniferous woodland, an estimated 3,221 tonnes of carbon could be captured and stored per year, over a period of 40 years (Table A. 8). This number accounts for 50% of the carbon emissions (currently 6,406 tonnes CO_2 equivalents-C per year) caused by the academic activities at the university (Newcastle University, 2021). Converting Nafferton Farm into a forestry research centre with mixed woodland (i.e. 50% coniferous woodland and 50% broadleaved woodland) could off-set 29% of these carbon emissions (Scenario 2; Table A. 8). Alternatively, 64% of these carbon emissions could be offset over a shorter period of about 5 years across the 2 farms, if the land use split increased the proportion of permanent grassland to what it used to be around 1900, as illustrated on an old map of Cockle Park Farm (Shiel, 2000) (Scenario 3; Table A. 10). Finally, by converting at each farm 81.5 ha of the arable land (29% at Cockle Park Farm and 17% at Nafferton Farm) into mixed woodland, 10% of these carbon emissions could be offset over a period of 40 years (Table A. 11).

The outcome of our interview with the farm manager regarding the difficulties of land conversion is listed in **Appendix D**. The soil carbon sequestration approaches discussed included altering the cultivation system, converting arable areas back to permanent grasslands or woodlands, and biochar application. From the feedback, various challenges exist, where the two main concerns of the farm manager were the restrictions of the tenancy contract and changes that might affect farm subsidies or tax status. When discussing farm-produced biochar as a carbon sequestration opportunity, it was stated that there were insufficient crop residues being harvested at the two farms, so one would need to purchase biochar from external providers to augment soil carbon, which would increase procurement costs. There were also concerns about the labour hours needed for spreading biochar and inspecting the soil health after adding biochar.

3.4. Discussion

Our survey of sixteen carbon management plans from academic institutions in and beyond UK found that none considered the terrestrial or soil carbon of their estates in a quantitative way (Table A. 1 **in Appendix A1**). Step 1 of the proposed methodological framework **(**Figure 3. 1) therefore sought to quantify the amount of carbon in the soil and tree biomass using Newcastle University's rural estate as an example, and it was found that the carbon in soil and tree biomass at its two research farms amounted to sixteen years of institutional carbon emissions. Hence, preserving or augmenting the terrestrial carbon of its estate is quantitatively important for Newcastle University's institutional carbon management plan. Considering that many other academic institutions in the UK and beyond have larger land holdings than Newcastle University (Barbiroglio, 2018), such findings are of broader significance.

The field work created a valuable database for studying relationships between current and past land use and terrestrial carbon stores (Step 2 in Figure 3. 1). Carbon density in soil was found to be dependent on soil collection depth, different farm locations and land use (woodland versus agricultural land, permanent grassland versus arable land at Cockle Park Farm), whereas the woodland vegetation, when the woodlands were established, and conventional versus organic management practices at Nafferton Farm, had statistically insignificant effects on soil carbon in our dataset. The finding that soil carbon storage in arable land were smaller than those in woodlands is in accordance with other studies (Reynolds et al., 2013, Wang et al., 2019). The negative correlation between soil pH and soil organic carbon storage which was found for the three soil depths at the two farms, is also in line with previous reports (Minasny et al., 2017, Reynolds et al., 2013). Soil pH is a primary control in environmental microbiology, and microbial processes, including the breakdown of organic matter into CO_2, are slowed down in acidic conditions, while soil pH also controls the carbonate equilibrium (Wiesmeier et al., 2019). Therefore, the relationship between soil pH and carbon content is often found to be significant (Reynolds et al., 2013). Agricultural land in the two farms were not subject to much liming and no buffering areas existed. The observed lower soil pH under woodland trees as compared to arable land may have contributed to the slower decomposition of soil organic carbon in undisturbed soil (Heikki Martti et al., 2016).

The TC density in the top soil layer (0-30 cm) at arable land across Cockle Park Farm and Nafferton Farm ranged from 18.26±4.82 $Kg\cdot m^{-3}$ to 23.08±4.96 $Kg\cdot m^{-3}$, respectively, which are lower than the average value (31.53 $Kg\cdot m^{-3}$) of a 0-15 cm arable soil survey over Great Britain (GB) (Reynolds et al., 2013). The mean TOC density over the top 90 cm soil layer in this study (Cockle Park Farm, 12.37±8.1 $Kg\cdot m^{-3}$; Nafferton Farm, 13.74±8.99 $Kg\cdot m^{-3}$) is lower than the average TOC of woodland, grassland and arable soil (17.56 $Kg\cdot m^{-3}$) up to 1 m depth across GB, but similar to the average TOC (14 $Kg\cdot m^{-3}$) in England alone (Bradley et al., 2005). At Nafferton Farm, we found similar soil carbon results to Zani et al. (2020) even though the soil sample processing steps and soil carbon determination method differed slightly between the two studies. One reason contributing to the greater amount of carbon of GB soils overall as compared to our results is the occurrence of peat-dominated soils in Wales and Scotland due to higher rainfall, which facilitates carbon sequestration (Balasubramanian et al., 2020, Guo and Gifford, 2002). Average rainfall from 2015-2019 was 1147 mm in GB, 902 mm in north-eastern England, 1560 mm in Scotland, and 1461 mm in Wales, respectively (MetOffice). When considering the impact of vegetation types within crop fields, our findings showed only minor effects on soil carbon density that are consistent with those of Badagliacca et al. (2018), whereas Wang and Sainju (2014) found that soil carbon is influenced by crop species. In this study, the division of Nafferton Farm into an organic part and conventional part over fifteen years from 2002 to 2017 had left no significant signature in soil carbon density (afterwards all the land was managed conventionally; samples were collected in March 2019). These results differed from those of Gardi et al. (2016) who stated that the soil carbon density would differ for various farming methods. We found higher mean carbon density in the 0-30 cm soil layer of permanent grassland as compared to arable land at both farms, which is consistent with the findings of Balasubramanian et al. (2020) and Gardi et al. (2016). Mean soil carbon density in coniferous woodlands were slightly, but not statistically significantly, higher compared to broadleaved woodlands on the two farms in this study, and this was also observed for Scottish forest soils (Vanguelova et al., 2013), forest soils in Great Britain (Reynolds et al., 2013), and parkland soils in southern Finland (Heikki Martti et al., 2016). Soil carbon storage for topsoil (0-30 cm) for both broadleaved (mean: 7.82 $Kg\cdot m^{-2}$) and coniferous woodlands (mean: 8.98 $Kg\cdot m^{-2}$) were slightly higher than the results obtained from the 2007 UK Countryside Survey (CS 2007, at 7.30 $Kg\cdot m^{-2}$ and 8.14 $Kg\cdot m^{-2}$, respectively) (Chamberlain et al., 2010, Reynolds et al., 2013). Vanguelova et al. (2013) explained such differences can be due to differences in soil bulk density. A lower bulk density value was used for CS 2007 (0.78 $g\cdot cm^{-3}$ for broadleaved and 0.52 $g\cdot cm^{-3}$ for coniferous) than in this research, which calculated the average bulk density among the three soil layers in woodlands as 0.81 g·cm^{-3} for Cockle Park Farm woodland soil and 0.86 g·cm^{-3} for Nafferton Farm woodland soil, respectively. Even if the difference was not statistically significant, an apparent increase in the mean carbon density of the surface soil of woodland at Cockle Park Farm, when comparing the longer established woodland with the more recently established woodland, is of interest. The difference observed is comparable to the range of several studies investigating soil carbon in relation to tree age (Hale et al., 2019, Heikki Martti et al., 2016, Vanguelova et al., 2013). Nevertheless, Hale (2015) also suggested that there was no systematic difference in soil properties between younger and older woodland growth. It should be noted that, even though most of the woodland in this study was over 100 years old, the trees were typically younger, while the soil carbon stock will have accumulated for the life of the woodland, not the individual trees.

Mean biomass C storage (Kg·m^{-2}) and stocks (tonnes) obtained from i-Tree Eco for the 40-year-old trees in this study (Table 3. 3) were comparable with those of mixed unmanaged growth stands in eastern Wales (65-year-old trees: 7.72-10.65 Kg·m^{-2}; ⩾65-year-old trees: 14.09-20.24 Kg·m^{-2}) (Hale et al., 2019). Only minor differences of tree carbon stocks were observed between the i-Tree Eco and equations in the Carbon Assessment Protocol of the Forestry Commission of England (Woodland Carbon Code). Since i-Tree Eco can be downloaded to smartphones, it enables raw data input directly from the field which is an attractive feature for carbon surveyors.

While the terrestrial carbon findings of this study overall were in good qualitative agreement with the wider literature, the field work established more reliable soil and tree biomass carbon data than could have been inferred from the literature. This is because land management effects on terrestrial carbon will depend on the local climate and geography. In addition, an analysis of the local land use history as part of step 2 of the proposed methodology (Figure 3. 1) brings the impacts of land management on terrestrial carbon stores closer to home. For example, at Nafferton Farm, the boundary between woodland and fields followed the contours of the Whittle Burn dene, which suggests that the local land use pattern may have resulted from mediaeval slash and burn agriculture, when the fields were created and woodland remained only in the most inaccessible areas (Ross, 2020). From the field data, this mediaeval conversion of woodland into agricultural land would have resulted in a terrestrial carbon loss of 14.45 Kg·m^{-2}. At Cockle Park Farm, an old map from ~1900 (Shiel, 2000) showed that 84% of the farmland was then managed as grassland, and only 16% as arable fields, versus 21% and 79% based on the recent records. According to the field data, this land use change resulted in a carbon loss of 3,251 tonnes from the terrestrial carbon stock while Cockle Park Farm was owned and managed by Newcastle University.

Based on analysis of terrestrial carbon stores for current and past land use, one can build realistic proposals for future land use change to augment the terrestrial carbon stores and thus to off-set institutional carbon emissions (CO_2 equivalents-C) (Step 3 in Figure 3. 1). For example, it becomes quickly apparent from Table A. 8-11, which predict the terrestrial carbon stocks as a function of land use on the estate, that substantial land use changes are required to off-set a tangible proportion of Newcastle University's current carbon emissions (CO_2 equivalents-C) over the next 40 years. According to our research findings, the most effective change to the land management regime would be to convert arable land into new grassland or woodland to sequester carbon in both soils and tree biomass, which is in line with the findings of other studies (Guo and Gifford, 2002, Hallsworth and Thomson, 2017, Rees et al., 2018). Carbon sequestration by converting agricultural land to woodlands has been discussed by several other authors (Kaplan et al., 2012, Minasny et al., 2017, Rees et al., 2018). In the UK, the annual amount of carbon removal from land conversion to forestry is 62% higher than for conversion to grassland (Hallsworth and Thomson, 2017). On average, the carbon storage in vegetation is lower compared with that in soil (Scharlemann et al., 2014) but mature woods are able to sequester considerably greater carbon than soil does (Hale, 2015).

Except for the conventional land use transformation, other approaches in terms of increasing soil carbon sequestration are worth researching. According to two long-term experiments at different locations in the south-eastern UK, with a duration of around 160 years, adding farmyard manure can achieve substantial accumulation of soil organic carbon (Poulton et al., 2018). Furthermore, 24 different long-term experiments across southern England over a period of 10 years showed how the application of various organic amendments (e.g. vegetable compost, sewage sludge) increased soil organic carbon at 23 sites (Poulton et al., 2018). Peatland is an ecosystem with the highest carbon density in the terrestrial environment (IUCN, 2018). Peatland covers 10 % of the UK land area, and the UK government has taken action on peatland restoration and preservation by sustainable management to maintain or improve the imperative role of peatland in carbon sequestration (IUCN, 2018). Besides, removing CO_2 by enhanced silicate rock weathering in croplands by introducing more base cations and a higher alkalinity environment, is an attractive technology because of its auxiliary improvement of crop productivity and agricultural soil properties (Beerling et al., 2020). In addition to enhancing the carbon stocks, soil management also needs to consider other important soil characteristics which include the water holding capacity, effect on nutrients, acidification risks (Scharlemann et al., 2014), indirect environmental impacts, and financial factors such as labour costs, loss of revenue from crops, the expenditure of purchasing saplings for woodland establishment and the expense of maintenance work. At Newcastle University, although the two farms are not yet part of the institutional carbon management plan, the farm director has developed his own carbon strategy and applied diverse carbon calculation tools to assess their current operations. Moreover, the farm managers would like to make more attempts, in cooperation with other departments or companies, to contribute more on carbon abatement (Questionnaire notes in **Appendix D**), whilst also voicing a number of concerns. Likewise, Aggarwal (2020) has debated the various difficulties on implementing a forest carbon project in northern India, involving eight villages with 107 households, and the dominant driving force leading farmers to withdraw from the project was the lack of economic gain. While it is thus acknowledged that land management decisions are made based on multiple additional criteria, the methodological framework developed in this study will help institutions to robustly consider in such decisions the implications on the terrestrial carbon in their estates.

3.5. Conclusion

This chapter demonstrated that carbon stocks in institutionally owned land can be substantial. Across the top 90 cm soil layer at two farms, woodlands soil TC densities were higher compared to agricultural land. In addition, woodland tree biomass carbon storage were 11.67 $Kg \cdot m^{-2}$ at Cockle Park Farm and 12.60 $Kg \cdot m^{-2}$ at Nafferton Farm. For the example of Newcastle University, the current carbon stock at its two research farms was 103, 620 tonnes in total, equivalent to sixteen years of institutional carbon emissions at the current rate (6,406 tonnes CO_2 equivalents-C per year). By converting 81.5 ha arable fields to mixed woodlands (half coniferous and half broadleaved trees) at each farm, 10% of these carbon emissions could be offset over the next 40 years. Various public and private sector institutions have very substantial land ownership and should consider the climate emergency when planning the way in which they manage their land. This chapter has developed a framework to derive a terrestrial carbon stock estimation by using field surveys, laboratory measurements and ecosystem modelling resources. The methodical framework which has been developed here can provide a perspective to researchers and executives on the realistic scale of updated carbon management plans, which not only quantify current and previous carbon stocks in institutionally managed land, but also consider potential realistic strategies to augment terrestrial carbon stocks in the green space under their management. Meanwhile the discussions with farm managers reveal an urgent need for more alternatives to increase soil carbon accrual with a range of agricultural carbon abatement practices such as no-tillage, the recycling of organic fertilizers, the application of soil amendments, best management strategies (e.g., high- productivity cultivars with increased plant density), enhanced silicate rock weathering in farming regions, and peatland restoration.

4. Chapter Four. Carbon stock of urban greenspace soils and plants on the main campus of Newcastle University

4.1 Introduction

Urban ecosystems represent a significant terrestrial carbon pool (Pouyat et al., 2002). In the UK, urban areas extend to over 1.8×10^{6} hectares and represent an estimated 8% of total land area (Office for National Statistics, 2019), which will increase further in the future due to urbanization. A total of 1.06 Mt (Megatonne) of carbon is reportedly stored in residential and non- residential land (0-100 cm depth soils with vegetation) in the city of Leicester in the UK. In the city of Bristol, UK, an estimated 0.098 Mt of carbon is stored in 618, 800 trees (i-Tree Bristol, 2019). Meanwhile, Wilkes et al. (2018) has emphasized that trees in inner west London can provide a similar aboveground biomass density as tropical forests. The amount of carbon removed by woodland in urban areas across the UK in 2017 was estimated to represent a value of £89 million (Office for National Statistics, 2019). Along with carbon sequestration, important ecosystem functions of urban greenspace include stormwater drainage, mitigating the urban heat island effect, environmental amelioration, air pollution reduction (Office for National Statistics, 2019, Edmondson et al., 2012, Hand et al., 2019), noise mitigation, improved citizens well-being and higher biodiversity (Edmondson et al., 2014). However, the increasing urban sprawl and transformation of landscapes may also impair ecosystem functions, and disturb the carbon cycle in natural soils (Richter et al., 2020).

The planting and management of urban greenspace will affect soil carbon concentrations (Lindén et al., 2020), and this provides a carbon sequestration opportunity. Climate, geological features, and the surrounding environments are statistically associated with urban forestry development (Hand et al., 2019, Heusinkvelt, 2016, Limoges et al., 2018, Viherä-Aarnio and Velling, 2017), and influence the ecosystem's capacity for carbon sequestration. Tree deaths resulting from planting the inappropriate trees exceed the sum of other insect- and diseased-related mortality (International Society of Arboriculture, 2020, Morani et al., 2011), which hampers efforts to store carbon into urban planted woodlands.

Many large organizations like universities have declared a climate emergency, and thus take their responsibility to mitigate climate change seriously. Such declarations by universities are important, because there is a lot of energy consumption and waste production from academic activities (De Villiers et al., 2014). Since many universities are located within cities, purposeful management of their greenspace to maximize carbon storage can set an example for these cities and their urban ecosystems (Cox, 2012, De Villiers et al., 2014, Wasikowski, 2017). There is often plenty of data available for the carbon emissions of universities and also information about the locations and species distribution in urban green spaces managed by universities, but the related carbon pool is rarely quantified and purposefully managed (Wasikowski, 2017). By considering augmentation of the carbon storage of trees and soil in greenspace on their campuses, universities and similar organizations can strengthen their carbon emission mitigation plans (De Villiers et al., 2014).

Only a few studies have investigated the carbon stock of trees owned by universities, and have calculated the related potential for offsetting university carbon emissions (Cox, 2012, De Villiers et al., 2014, Sharma et al., 2020). In the UK, only the University of Leeds has quantified that 540 tonnes of carbon are stored in 1,450 trees on its campus (Gugan et al., 2019). In New Zealand, De Villiers et al. (2014) have calculated that the 4,137 campus trees on a university campus in New Zealand can store a total of 1,585 tonnes of carbon. Cox (2012) estimated that the total carbon content of all trees on a California State University campus was 862 tonnes. Across 24 hectares of urban campus of Amity University in India, totally 1,997 trees from 45 different tree species presented a C pool of 140 tonnes (Sharma et al., 2020). However, these works examined tree carbon storage without consideration of the related soil carbon storage.

Newcastle University declared a climate emergency in 2019 and aims to achieve net-zero carbon by 2030 (Newcastle University, 2021), as does Newcastle City Council (Newcastle City Council, 2020). The university currently does not know nor actively manage the carbon stored in the soils and trees of its urban campus as part of its carbon management plan. The main objectives of this chapter were therefore to i) quantify the soil and tree carbon across the greenspace of the urban Newcastle University campus to produce a terrestrial carbon storage baseline; ii) to review the species selection, planting patterns and growth status of trees currently on campus with a view of optimizing their carbon storage potential; iii) to obtain from interviews with university estates and sustainability managers of the university an understanding of the challenges in implementing institutional plans to enhance the carbon stock of urban greenspace; iv) to consider how lessons learned could be applied more widely by the local council for terrestrial carbon management and off-setting at the city-scale. This is the first study of its type that integrates soil and biomass carbon in urban greenspace with institutional net-zero carbon aims and presents an approach that can be adopted widely and internationally.

4.2 Materials and methods

4.2.1 Study areas

Newcastle is in north-eastern England with a population of around 320,000 (Population UK), and responsible for 335,400 tonnes of carbon emissions in 2019 (National Atmospheric Emissions Inventory). The city experiences a temperate oceanic climate characterized by a slightly hot and dry summer (average 13 °C), a cloudy and wet winter (average 5 °C), and 902 mm rainfall each year (MetOffice). Where natural soils are present, soil texture within the region is dominated by loamy and clayey soil (http://www.landis.org.uk/soilscapes/), although much of the city centre is built on soils that have been disturbed by hundreds of years of construction and demolition. Newcastle University is a public research university with 3,500 staff and 28,000 students, and was responsible for 6,406 tonnes CO_2-equivalent C of greenhouse gas emissions in the academic year 2019/20 (Newcastle University, 2021). Newcastle University established its main campus in 1834 in Newcastle city centre, which nowadays extends to 25 hectares and accounts for 1.4 % of the urban greenspace areas (Table B. 1 **in Appendix B**). Its Heaton Sports Ground has been used for sports since at least the 1890s (Digimap, 2020), the topsoil being managed using specialist sands to produce a turf playing surface suitable for cricket, rugby and football.

4.2.2 Soil Survey

In September 2020, soil sampling was performed across the greenspace of the main university campus and a suburban university sports area in the city centre of Newcastle (see Figure 4. 1). The top 0-30 cm of the soil profile were sampled as they tend to contain the most carbon and were also unlikely to contain services (pipes and cables) that could have been damaged by the sampling. Previous research recommended that 30-50 soil sampling points should provide a reliable representation of soil carbon for different land covers (Edmondson et al., 2014). Therefore, 42 locations were randomly generated across three land-use classes: 13 sampling points were within a small campus woodland/park (0.2 hectares), 12 sampling points were across lawned areas (0.3 hectares) with some free-standing trees of the central campus, and 17 sampling points were at the University's Heaton Sports Ground. The collection and carbon measurement methods and the related carbon content calculation of soil samples as well as the methodology of pH and X-ray diffraction analysis can be found in **Appendix B1**.

Figure 4. 1 The location of the study areas. a) the location of Newcastle upon Tyne in the UK; b) study area locations in Newcastle upon Tyne; c) the area of the Heaton Sports Ground managed by Newcastle University; d) the central campus of Newcastle University.

4.2.3 Tree carbon quantification

The tree database for the main campus of Newcastle University was obtained from the estate support management office in July 2019 and contained 490 trees, including 473 free-standing trees and 17 small groups of trees (i.e. dense areas where trees grow extremely close together). A related report summarizing the tree species, risk levels, health conditions and life expectancy of trees with management suggestions for the main campus of Newcastle University, was provided by the Estate office of the university. This report was compiled by several invited arborist consultants with the visual assessment based on the qualified tree assessment guidance (Bethge and Mattheck, 1993, Lonsdale, 1999, Matheny, 1994). The diameter at breast height 1.3 m above ground (DBH) was measured using a rounded down diameter tape; the height of trees was visually estimated by the arborists; the possible canopy was obtained using either a tape or measuring wheel and an estimation was given where the site access was restricted.

Two methods were used to estimate the carbon stored by the trees listed in the database: (1) i-Tree Eco; (2) allometric biomass equations. i-Tree Eco is one programme package in i-Tree tools (https://www.itreetools.org/), developed by the United States Department of Agriculture Forest Service, which can effectively assess the benefits of green space and quantify the structure of community trees and has been conducted over lots of countries. The related tree biomass equations for each species were found from the literature. When applying these two approaches, if no specific plant category can be matched for a tree species or no biomass formula was found for a particular species, the tree species would be assigned to one species group which came from the most closely related family. As for group trees, due to the difficulty in estimating the specific physical parameters of each tree species, a general tree aboveground biomass equation from Jenkins et al. (2003) was substituted. The details of how to upload tree parameters to i-Tree Eco and the calculations regarding total tree carbon stock (Kg), tree canopy cover (m^2) and tree carbon storage density ($Kg \cdot m^{-2}$) are provided as **Appendix B2**. Results of allometric biomass equations were included in the paper, while the output of i-Tree Eco was added to the **Appendix B2**.

Considering that many tree species in the database contributed only a single tree on campus, tree characteristics of these plants were less likely to be representative of general growth conditions for that tree species in urban areas in north-eastern England. Therefore, this chapter only considered the eight largest groups in terms of tree numbers to statistically analyse the variations of life stage, DBH, tree height, tree canopy cover and carbon content between various tree groups and within a group of an individual tree species.

4.2.4 Questionnaire design for interviewing the sustainable campus and estate managers

We drew on three sources of information to understand the land management and how it may relate to the climate action plan of Newcastle University, based on the interview with the estate manager who designs the campus greenspace, and two team members who frame the carbon management plan. In the questionnaire, we presented data obtained from the chapter 3 which was based on the same aim to offset institutional carbon emissions, and proposed several possible options to improve terrestrial carbon sequestration in Newcastle University's urban campus, and then inquired about any concerns in terms of acceptability that the managers had for the approaches we mentioned. The questions for each interviewee are attached in **Appendix D**. Also included in the analysis were interviews undertaken by a master dissertation project of Newcastle University (Ross, 2020) which asked representatives from 27 city councils across the UK to state the opinions of their councils in terms of tree planting for alleviating climate change.

4.2.5 Data analysis

ArcGIS (version 10.6.1) was used to produce the maps with labelled soil carbon content values. The effects of various land cover classes or sampling locations on soil carbon storage and soil pH were analysed using multivariate analysis. Pearson correlation was applied to access the relationship between soil pH and total soil carbon, organic carbon, and inorganic carbon, respectively. The statistical relationship between tree carbon stock and tree age was determined by one-way ANOVA. Both were processed by SPSS (IBM SPSS Statistics 26, USA). Statistical significance is acknowledged as $p \leq 0.05$.

4.3 Results

4.3.1 Carbon storage and mineral compositions of urban topsoil in campus green space

The carbon storage ($Kg \cdot m^{-2}$) for the 42 urban campus soil samples at 0-30 cm depth is shown in Table 4.1. To visualise these results, Figure B. 1-3 **in Appendix B** show the distribution of soil collection points and the corresponding specific soil carbon values. The bulk density in the study sites was 0.77±0.1 $g \cdot cm^{-3}$ in Heaton Sports Ground, 0.87± 0.08 $g \cdot cm^{-3}$ in campus lawn, and 0.83± 0.08 $g \cdot cm^{-3}$ in campus woodland. These data were all similar to the soil density recorded in other green areas from the same region (UK Soil Observatory, 2021). The average total soil carbon (STC), organic carbon (SOC) and inorganic carbon (SIC) values for the whole institutional land are 18.85± 6.34 $Kg \cdot m^{-2}$, 13.52± 4.23 $Kg \cdot m^{-2}$ and 5.33± 2.81 $Kg \cdot m^{-2}$, respectively. STC in the urban campus woodland park (23.05± 6.43 $Kg \cdot m^{-2}$) and lawned areas with free-standing trees (22.29±4.57 $Kg \cdot m^{-2}$) presented a significantly larger carbon content than the suburban sports field (13.25±1.65 $Kg \cdot m^{-2}$) (One-way ANOVA, $p<0.001$, Table 4.1). In addition, for the proportion of SOC to STC, the data in Heaton Sports Ground (76%) was higher than urban lawned and woodland parks (both are 70%). Additionally, compared to Heaton Sports areas, soil pH obtained in the urban campus was higher (One-way ANOVA, $p<0.001$, Table 4.1). All soil carbon types significantly increased with increasing soil pH, although this relationship was less obvious in SOC (Pearson correlation, $p<0.05$, Table B. 4).

	Number of soil samples	Soil bulk density ($g \cdot cm^{-3}$)		STC ($Kg \cdot m^{-2}$)		SOC ($Kg \cdot m^{-2}$)		SIC ($Kg \cdot m^{-2}$)		Soil organic carbon %	Soil inorganic carbon %	Soil pH	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD			Mean	SD
Heaton Sports Ground	17	0.77	0.10	13.25	1.65	10.07	1.66	3.18	0.79	76%	24%	6.48	0.60
Urban lawn with some free-standing trees	12	0.87	0.08	22.29	4.57	15.59	4.18	6.7	3.41	70%	30%	7.32	0.77
Urban woodland park	13	0.83	0.08	23.05	6.43	16.13	3.58	6.92	1.89	70%	30%	7.45	0.38
Total	42			18.85	6.34	13.52	4.23	5.33	2.81	72%	28%	7.02	0.74
Significant difference between different land uses				<0.001		<0.001		<0.001				<0.001	

Table 4. 1 Bulk density ($g \cdot cm^{-3}$), carbon content ($Kg \cdot m^{-2}$) and pH of the 0-30 cm profile of urban greenspace soils with the statistical significance for soil carbon caused by different land uses.
STC: soil total carbon; SOC: soil organic carbon; SIC: soil inorganic carbon. SD: Standard deviation. Significant (p<0.05) findings have been recognized.

X-ray diffraction patterns of soil samples are displayed in Figure 4. 2. XRD diffractograms of soil samples in this study all showed the major peaks of quartz at similar positions (2θ=20.8º and 2θ=26º). In all 10 selected samples calcite (the major mineral host for inorganic carbon) was only identified in two urban woodland soils (a, b, in Figure 4. 2). Kaolinite, a clay mineral, was mainly found in urban campus samples but not at Heaton Sports Ground. Additionally, orthoclase, a type of potassium feldspar, was present in small amounts in soils from both locations.

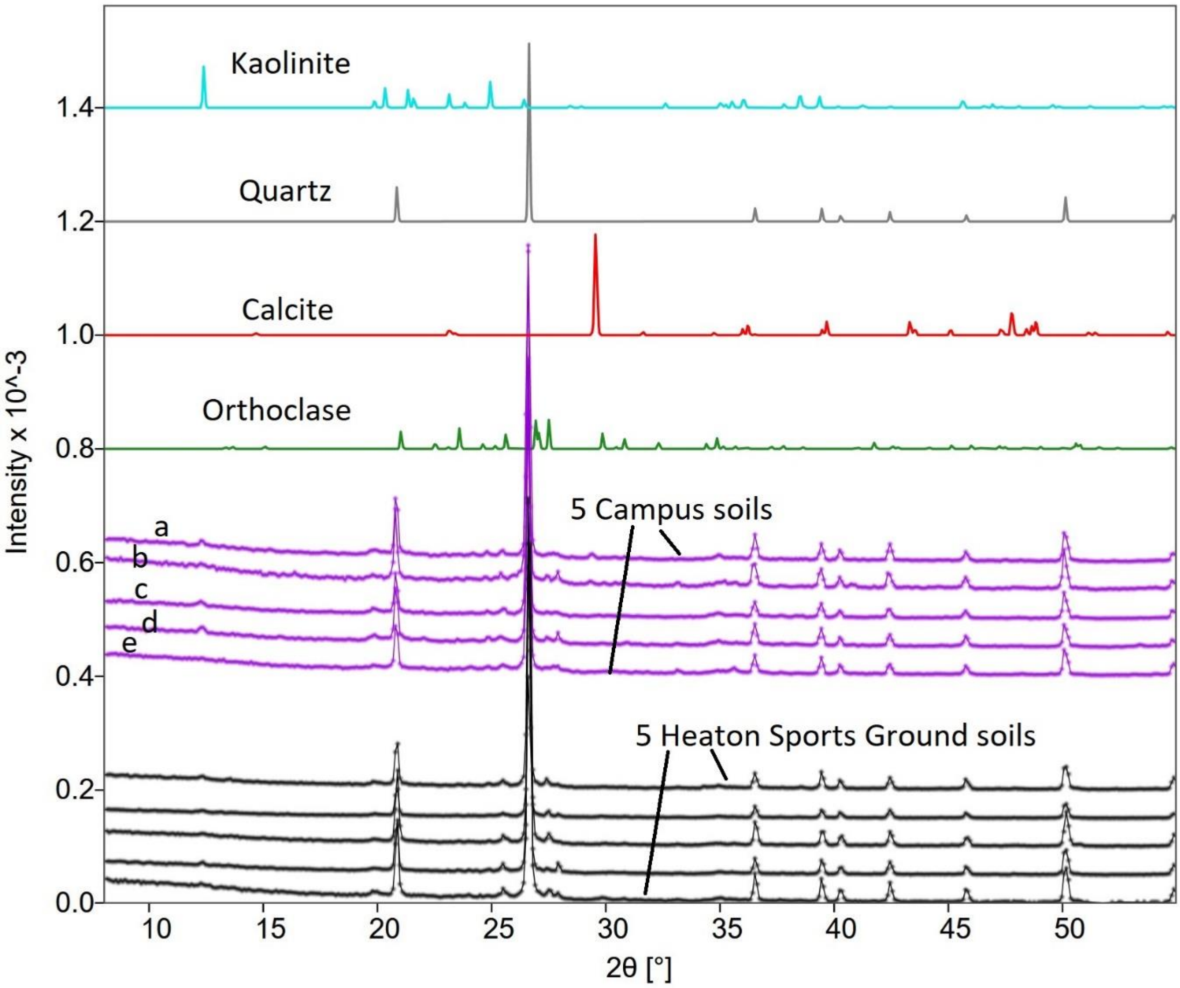

Figure 4. 2. X-ray diffraction patterns of 10 selected soil samples.
The top 4 spectra are XRD patterns of reference minerals. The 5 purple XRD patterns show the soils collected in the urban campus of Newcastle University: a&b from the woodland; c&d&e from the urban lawns with free-standing trees. The 5 black patterns at the bottom are the results of soils from Heaton Sports Ground.

4.3.2 Carbon stock of all trees on the central campus and a comparison of two methods of tree carbon storage estimation

Altogether 473 individual trees and 17 tree groups on Newcastle University campus were analysed, covering 67 tree species, and their allometric aboveground biomass equations and the number of trees per species are included in Table B. 2**,** citing literature source for equation coefficients. Across the central campus, of twenty-three tree species only one tree was found in each species; in thirty-eight tree species the number of trees ranged from 2-20 and the population of the remaining six tree species were all above 20. Over 20% of the total tree population was accounted by Large-Leaved Lime (*Tilia platyphyllos*), making lime the most prevalent campus tree in Newcastle (Richter et al., 2020).

DBH is the most common variable to calculate aboveground tree biomass using allometric models. Overall, estimates of the entire tree canopy cover in the central campus of Newcastle University was 2.92 ha, measured by allometric biomass equations, corresponding to the total carbon stock from all trees of 223.5 tonnes (Table 4.2). Meanwhile, the average tree carbon storage was 76.6 tonnes per hectare of canopy by applying empirical biomass formulas (Table 4.2).

The carbon storage from individual tree species using the two methods is summarised in Figure B.4 with the Pearson correlation between the two methods. The number of tree species in Figure B.4 is fewer than Table B. 2, because some classifying categories of tree species in the i-Tree Eco database which the programme attributed automatically are different from the literature data selected by the authors. To clarify and simplify the data presentation, Figure B.4 only lists the tree species which belong to the same plant family between the two approaches. Across the 46 tree species, the trendline explained a high proportion of the correlation between i-Tree Eco and allometric biomass equations when evaluating tree carbon storage (R^2=0.9337, Figure B.4). From here on, the tree carbon values given in the following sections are all calculated from allometric biomass equations to avoid the confusion when two sets of data are presented at the same time.

The carbon stock in 67 tree species calculated using allometric biomass equations is summarised in Figure 4.3. Large-Leaved Lime is the numerically dominant tree species from the survey across Newcastle University. The carbon stock from Large-Leaved Lime (40,721 kg) and Sycamore (*Acer pseudoplatanus*) (40,238 kg) was similar, where both together accounted for the largest percentage of carbon stock from all trees on campus (36.2%), although the number of Large-Leaved Lime (95) was almost double the number of Sycamore (54) (Figure 4.4 a). Twenty-nine Ash (*Fraxinus excelsior*) and seventeen Swedish Whitebeam (*Sorbus intermedia*) were the third (27,713 kg) and fourth group (16,756 kg) in terms of the total carbon stock per tree species. Beech (*Fagus sylvatica*), London Plane (*Platanus hispanica*), Kanzan Cherry (*Prunus serrulata*), Cappadocian Maple (*Acer cappadocicum*), Norway Maple (*Acer platanoides*), Copper Beech (*Fagus sylvatica "Purpurea"*), and Sugar Maple (*Acer saccharum*) were tree species which each stored carbon ranging from 5-10 tonnes, and the carbon stored by the rest of species groups was less than 5 tonnes (Figure 4.3 b**).** The carbon stock per tree species varied widely, which became clear when comparing tree species with the same population. Although Beech (*Fagus sylvatica*), Common Oak (*Querous robur*), and Lawson Cypress (*Chamaecyparis lawsoniana*), each accounted for 0.8% of the total campus tree population, their carbon stock was 9,982 kg, 1,361 kg, and 854 kg, respectively (Figure 4.3**).**

Item	Unit	Data
Soil carbon storage in four Sports Grounds	$tonnes \cdot ha^{-1}$	132.50
Soil carbon storage in central campus	$tonnes \cdot ha^{-1}$	226.60
Soil carbon storage of all university-owned land (sports areas& central campus land)	$tonnes \cdot ha^{-1}$	188.50
Carbon storage of 490 trees calculated from biomass equations	tonnes per hectare canopy	76.6
Area of four Sports Grounds	hectares	16.00
Area of central campus	hectares	9.00
Area of all university-owned land (sports areas& central campus land)	hectares	25.00
Area of 490 trees canopy	hectares	2.92
Carbon stock in four Sports Grounds	tonnes	2120
Carbon stock in central campus	tonnes	2039
Carbon stock of all university-owned land (sports areas& central campus land)	tonnes	4159
Carbon stock of 490 trees calculated from biomass equations	tonnes	223
Total terrestrial carbon stock in the university (soils & trees)	**tonnes**	**4383**

Table 4. 2 Overview of carbon storage ($tonnes \cdot ha^{-1}$) from different components of campus greenspace and the total estimated carbon stocks (tonnes) over the greenspace of Newcastle University and the city of Newcastle upon Tyne.
[a]: Data sources of open space areas from Newcastle City Council (2018)
[b]: The tree cover of Newcastle is estimated as 18.1% (Newcastle City Council, 2019).

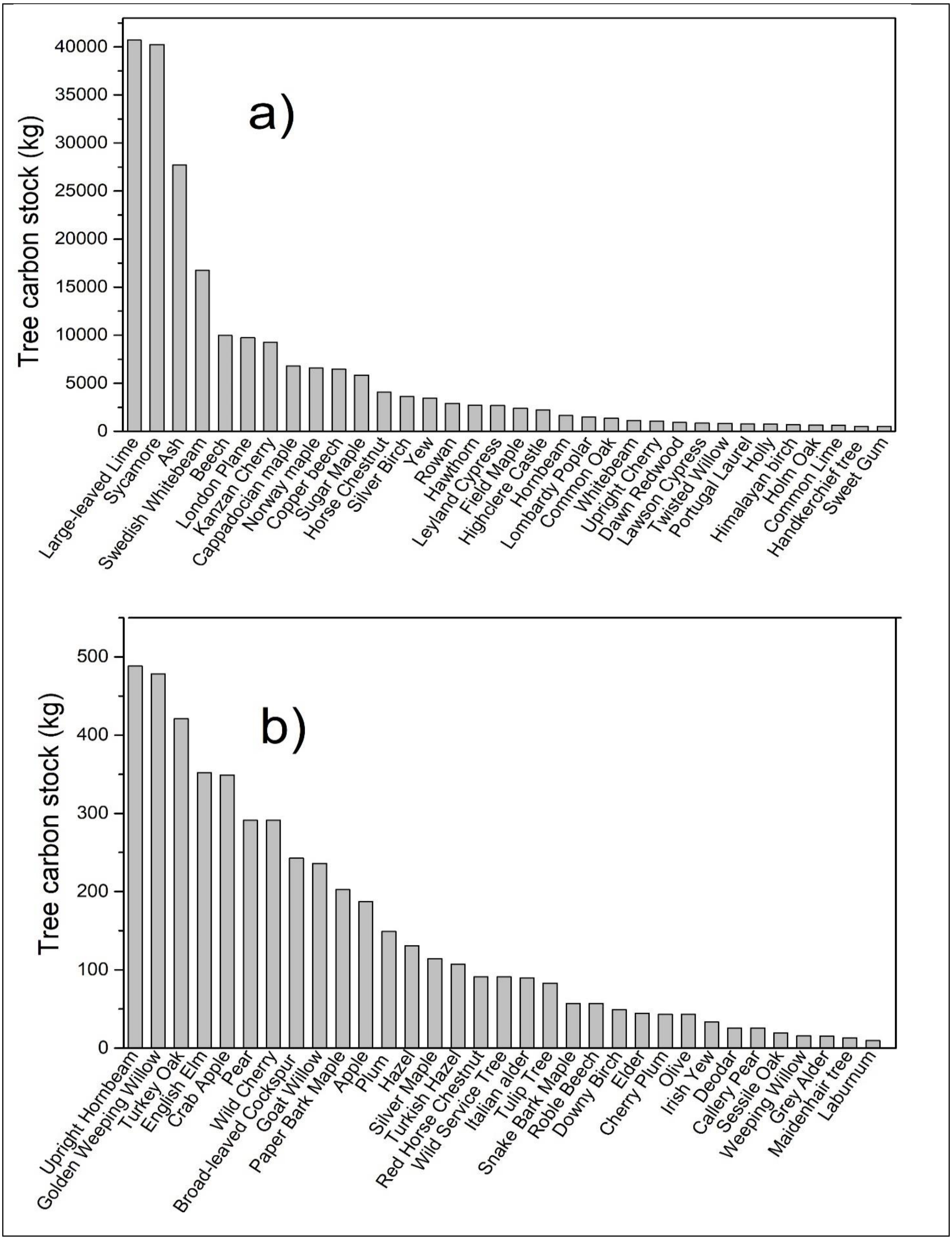

Figure 4. 3. Total carbon stock (kg) of all trees from each tree species on the campus of Newcastle University calculated by allometric biomass equations (overall 67 tree species). a): the tree species with the total carbon stock ≥ 500 kg; b): the tree species with the total carbon stock ≤500 kg.

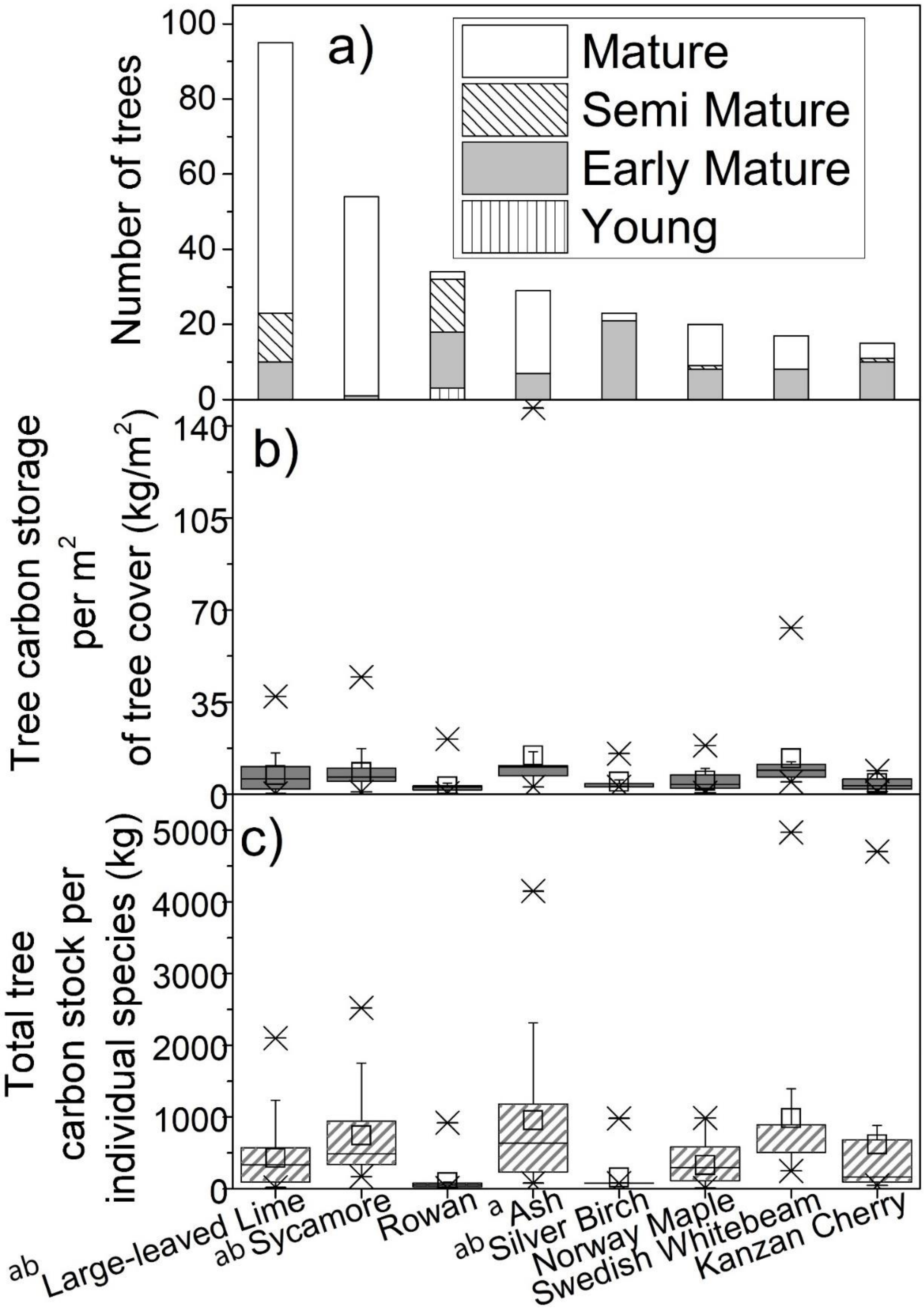

Figure 4. 4. a) The distribution of life stage per tree species among the eight species which occur most frequently on the campus of Newcastle University; b) the average carbon storage per m^2 of tree cover; c) the carbon stock per individual tree

[a]: tree carbon stock from an individual tree of this species is significantly impacted by the tree age;

[b]: tree carbon storage of this species is significantly impacted by the tree age.

4.3.3 The variation of tree characteristics and carbon stock from the eight largest groups in tree numbers

The eight largest tree groups in terms of population are Large-Leaved Lime (95, 20.1% of total), Sycamore (54, 11.4% of total), Rowan (*Sorbus aucuparia*) (34, 7.2% of total), Ash (29, 6.1% of total), Silver Birch (*Betula pendula*) (23, 4.9% of total), Norway Maple (20, 4.2% of total), Swedish Whitebeam (17, 3.6% of total), and Kanzan Cherry (15, 3.2% of total). The change of mean tree DBH, tree height and tree cover area with variation in tree age can be found in Table B. 3, and the distribution of life stage per tree species is displayed in Figure 4.4-a. There was a difference regarding tree age composition between various tree species in the urban campus. An assured planting date would be a great help to assess tree age, while other parameters of tree growth performance can assist to estimate the stage phase of trees, such as circumference of trunk, growing site conditions, size of buds, trunk colour, crown transparency, and loss/death of biomass, etc (ICP Forests, 2016, Ostberg et al., 2021). The latter method is how the arborists classified the maturity of campus trees in this study. Almost all of the Sycamore (98%) were mature trees, whereas just 6% of Rowan and 9% of Silver Birch were close to their fully mature stage. Semi-mature trees can be only found in Rowan (41%) and Large-Leaved Lime (14%), with relatively few in Kanzan Cherry (7%) and Norway Maple (5%). The remaining trees among eight tree species groups can be classified as early maturity trees, apart from 9% of Rowan recently planted which were not fully established yet. Over the eight different tree species, DBH, tree height and tree canopy area all increased with the increasing maturity of tree life stage in general, although there was a lack of a statistically significant difference among some tree species (Table B. 3). Only data from Ash and Silver Birch demonstrated that three tree dimensions would be significantly affected by tree ages in this report, while one or two plant physical parameters of Large-leaved Lime, Rowan and Norway Maple could show an important relationship with the tree mature stages (One-way ANOVA, $p \leq 0.05$, Table B. 3).

The average carbon content per square meter of tree cover from Ash, was the largest (14.67 $kg \cdot m^{-2}$), followed by Swedish Whitebeam (13.57 $kg \cdot m^{-2}$), Sycamore (8.36 $kg \cdot m^{-2}$), and Large-Leaved Lime (7.21 $kg \cdot m^{-2}$) (Figure 4.4 b). In contrast, considering the total carbon stock (kg) of each single tree from the eight tree species, the value from Swedish Whitebeam was the highest averaging 986 kg, which was slightly larger than Ash (956 kg). Furthermore, as a whole tree body, the single tree having the lowest ability for storing carbon was Rowan (88 kg) (Figure 4.4-c). Although the number of lime trees was the largest, the average tree carbon storage per square meter and carbon stock per individual tree of lime were not the greatest, which means that the dominant tree species does not necessarily store the most carbon under the average situation and vice versa (Richter et al., 2020). In general, the accumulation of carbon per square meter of tree cover and per individual tree from the plant species whose trees were mostly in the mature stage was higher than for younger trees. Trees from the same age group still showed a different carbon storage ability for different species. For instance, for those tree species contained the trees of semi-mature stage, Kanzan Cherry had the highest carbon storage (7.26 $kg \cdot m^{-2}$) while Rowan's was the lowest (1.87 $kg \cdot m^{-2}$); in the early mature stage, the carbon storage of Swedish Whitebeam was the top (7.29 $kg \cdot m^{-2}$), and Large-leaved Lime showed the lowest ability in carbon storage (2.5 $kg \cdot m^{-2}$); again, in the mature stage, the carbon storage ($kg \cdot m^{-2}$) was the highest for Swedish Whitebeam (17.96 $kg \cdot m^{-2}$), slightly higher than Ash (17.53 $kg \cdot m^{-2}$) and Silver Birch (15.17 $kg \cdot m^{-2}$), while mature Kanzan Cherry showed a much lower carbon storage ability (4.83 $kg \cdot m^{-2}$) (Table 4.3). Not all tree species could show a significantly statistical relationship between the biomass carbon index and growth stage. For example, Large-Leaved Lime, Rowan, and Silver Birch, had significantly higher carbon storage ($kg \cdot m^{-2}$), and mean carbon stock per individual tree (kg) with increasing tree age ($p \leq 0.05$, Figure 4.4b&c, Table 4.3). However, only one index from Ash -the mean carbon stock per individual tree (kg), significantly increased with the tree growth ($p \leq 0.05$, Figure 4.4 c, Table 4.3).

4.3.4 The entire carbon stock potential of the urban ecosystem

The estimated carbon stock from 0-30 cm topsoil and all trees in the central campus of Newcastle University and Newcastle City is summarised in **Table 4.2 and Table 4.4,** respectively, based on extrapolation of the campus data to larger city greenspace areas. The four sports grounds and other green spaces forming the urban campus of Newcastle University store 4,159 tonnes of soil carbon; if including the 223 tonnes of tree carbon, both contribute to the total greenspace carbon stock of 4,383 tonnes (175 $tonnes \cdot ha^{-1}$). This number is equivalent to 68% of the CO_2-C equivalents emitted in the university in 2019/20 (6,406 tonnes).

For each greenspace and land-cover class, the geology and history of land management, soil compaction, etc., will affect carbon storage, but an initial estimate can nonetheless be obtained by extrapolation of the Newcastle University city campus field data to the total urban greenspace of Newcastle City. Accordingly, a total of 397,648 tonnes soil carbon and 24,593 tonnes tree carbon are the estimated current carbon stock of Newcastle city greenspace (Table 4.4). As a conclusion, an estimated terrestrial carbon pool of 422,241 tonnes is claimed across the whole urban greenspace owned by Newcastle City Council, which is 26% greater than the total CO_2-C equivalents emissions of Newcastle City in 2019 (335,400 tonnes) (Table 4.4) (National Atmospheric Emissions Inventory, 2019).

We have introduced five scenarios (Table B. 5-8 in **Appendix B**) regarding the carbon stock potential due to future land conversion, accounting for the time needed for trees to become mature. The performance of carbon storage of trees varies for the different growth stages (Liepiņš et al., 2016), and this variation would differ between tree species, but for facilitating the calculation, this study estimated the time for trees to become fully mature as 57 years by referencing the relationship between tree age and DBH (Table B. 5-8 in **Appendix B**) (McPherson et al., 2016). Meanwhile, it was assumed that soil would also need 57 years to reach the present carbon equilibrium of each land cover class. Therefore, in the scenarios below, we divided the difference between the future and current carbon storage ability of each greenspace by 57 years to obtain the annual carbon capture and storage achieved with the land use change (tonnes·ha^{-1}·year^{-1}). The first scenario targets the university campus and the other three are for Newcastle City. Firstly, by converting all available green areas on campus to woodland containing the top 4 tree species with the highest carbon storage ability (Ash, Swedish Whitebeam, Sycamore, and Large-Leaved Lime; Figure. 4.4), the estimated annual increase in the greenspace carbon stock would compensate for only 1.13% of CO_2-C equivalents emissions of the university produced per year at the rate stated for 2019. In the same way, under scenario 2, if extending the woodland with these 4 tree species over the total urban greenspace, the additional carbon captured and stored annually would offset only 0.95% of the annual CO_2-C equivalents emissions of Newcastle City at the rate stated for 2019. For comparison, Newcastle City Council (2019) has targeted to increase urban tree cover of its greenspace from the current 18.1% to 20% by 2050, and thereby an extra 33.7 ha lands can be afforested. Either introducing only 4 tree species (Ash, Swedish Whitebeam, Large-Leaved Lime, Sycamore) or mixed woodland, the annual carbon captured and stored by the 33.7 ha of new urban woodland amounts for respectively, only 0.020% and 0.014% of the annual CO_2-C equivalents emissions of Newcastle City at the rate stated for 2019 (Scenarios 3&4). From these scenarios, we understand the current woodland expansion project only contributes a small amount towards city council carbon offsetting. For each land cover type, the soil carbon pool will reach a new equilibrium state between the accumulation of fresh carbon from decaying leaves/grass clippings, etc. and soil respiration of organic matter by bacteria, while new urban woodland would also enhance terrestrial carbon accumulation in the biomass of tree trunks and branches.

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	Carbon storage per m^2 of tree cover (kg·m^{-2})					**Carbon stock of individual tree (kg)**					**Statistical significance as the change of maturity (*p* value）**	
	Young	Semi Mature	Early Mature	Mature	Mean	Young	Semi Mature	Early Mature	Mature	Mean	Carbon storage per m^2 of tree cover (kg·m^{-2})	Carbon stock of individual tree (kg)
Large-leaved Lime	n.a.	2.44	2.5	8.73	7.21	n.a.	114.16	127.04	527.32	428.64	**0.001**	**<0.001**
SD	n.a.	2.37	0.7	7.34	6.98	n.a.	125.16	88.3	398.94	392.13		
Sycamore	n.a.	n.a.	4.94	8.49	8.36	n.a.	n.a.	387.56	759.33	745.56	0.471	0.377
SD	n.a.	n.a.	0	6.86	6.76	n.a.	n.a.	0	584.51	577.74		
Rowan	2.4	1.87	2.86	11.88	2.94	29.54	41.89	78.03	571.64	87.91	**<0.001**	**<0.001**
SD	1.54	0.92	1.33	12.79	3.4	0	22.49	49.72	493.37	155.11		
Ash	n.a.	n.a.	5.68	17.53	14.67	n.a.	n.a.	163.08	1207.8	955.63	0.299	**0.019**
SD	n.a.	n.a.	2.89	29.2	25.84	n.a.	n.a.	56.44	1096.71	1053.44		
Silver Birch	n.a.	n.a.	3.76	15.17	4.75	n.a.	n.a.	89.86	865.47	157.31	**<0.001**	**<0.001**
SD	n.a.	n.a.	0.8	0.37	3.38	n.a.	n.a.	26.51	164.07	227.58		
Norway Maple	n.a.	6.8	3.35	6.19	5.08	n.a.	768.58	192.79	389.08	329.54	0.356	0.101
SD	n.a.	n.a.	2.72	5.1	4.31	n.a.	n.a.	208.04	304.79	291.39		
Swedish Whitebeam	n.a.	n.a.	7.29	17.96	13.57	n.a.	n.a.	558.73	1284.51	985.66	0.136	0.189
SD	n.a.	n.a.	2.19	17.64	14.36	n.a.	n.a.	435.28	1333.97	1098.89		
Kanzan Cherry	n.a.	7.26	3.45	4.83	4.07	n.a.	684.39	309.1	1373.95	618.08	0.199	0.324
SD	n.a.	n.a.	1.56	3.22	2.23	n.a.	n.a.	312.59	2223.44	1163.58		

Table 4. 3 the eight most frequently occurring trees and their corresponding mean summary of carbon storage per m^2 of tree cover (kg·m^{-2}) and carbon stock of individual trees (kg) within the different age classifications and the statistical significance of the division by age groups. SD: Standard deviation. Significant findings have been highlighted (p<0.05); n.a: no data for that group

Item	Unit	Data
Soil carbon storage in sports grounds (referenced from the value obtained in Newcastle University)	tonnes·ha^{-1}	132.50
Soil carbon storage in other greenspace (referenced from the value obtained in Newcastle University)	tonnes·ha^{-1}	226.60
Urban trees carbon storage (referenced from the value obtained in Newcastle University)	tonnes per hectare canopy	76.6
Area of sport pitches managed by Newcastle Council [a]	hectares	46
Area of other greenspace managed by Newcastle Council [a]	hectares	1728
Area of urban trees [b]	hectares	321
Greenspace soil carbon stock	tonnes	397,648
Urban trees carbon stock	tonnes	24,593
Total terrestrial carbon stock from open green space (soils & trees)	**tonnes**	**422,241**

[a]: Date sources of open space areas from Newcastle City Council (2018) and the detailed value can be found in **Table B1** in **Appendix B**.
[b]: The current tree cover of Newcastle is estimated as 18.1% (Newcastle City Council, 2019).

Table 4. 4 The estimated carbon storage (tonnes·ha^{-1}) and the estimated total carbon stock (tonnes) over the city of Newcastle upon Tyne.

4.3.5 Questionnaire feedback from estate and carbon managers

From the interview with university managers, Table 4.5 summarised the factors driving tree species selection and planting in Newcastle University. The tree species selection is not currently based on augmenting terrestrial carbon, which is in line with the feedback from 27 UK city councils (20 in England, 3 in Scotland, 3 in North Ireland, 1 in Wales) (Ross, 2020), and other findings of previous studies (Heusinkvelt, 2016, Limoges et al., 2018, Morani et al., 2011, Sanders et al., 2013, Scholz et al., 2016). Both interviewees suggested a higher density of tree plantation is less achievable if the greenspace location is near a commercial centre with a built-up nature and complex belowground services. Particularly, the estate manager concerned the shape and canopy of fully grown trees might interfere with electricity wires or traffic (Spengler and Ellis, 2019). Also, the roots of mature trees could damage fundamental urban infrastructures such as water lines, and root heave, where trees' roots encroach the sidewalk or curbs to "escape" limited space or compacted soil conditions, leading a considerable repair cost (Randrup et al., 2001, Scholz et al., 2016). All these considerations are important issues to evaluate in urban tree planning (Sanders et al., 2013). More importantly, in urban settings, the availability of areas for tree planting is highly constrained (Ross, 2020). Indeed, the carbon storage capability of specific tree species is worth considering when designing a campus ecosystem, but balancing benefits from other research and business interests is inevitable (Table 4.5). This series of thinking has affirmed the predominant role of socioeconomic variables rather than biophysical variables in determining the carbon storage potential of urban greenspace on an institution-scale.

<table>
<tr><td>1. Factors influencing tree species selection</td><td>Ground manager:
• Being street replacement or not
• The distance of trees to buildings nearby
• Health condition of previous trees
• Being a memorial tree or not
• Size of planted ground
• Shape and crown of trees
• Survivability of trees at in-situ climate
• Trees possessing a longer growing seasons are preferred, i.e., cherry trees, fruit trees
• Popular tree types for gardening globally</td></tr>
<tr><td>2. Considerations when planting additional trees for carbon abatement by the institution</td><td>Ground manager:
• Limited plantation space because the central campus is close to city commercial centre
• Necessity to balance other performances the tree presents
• Possible shape of trees in the future
• Risk of disease spread among same tree species
Carbon& Energy manager:
• Performance of other functions from the trees at the same time, i.e., well-being benefits.
• Spatial constraints due to the compact nature of most of the central campus
• Whether being the best use of institutional resources
• Acceptability from other stakeholders in the university</td></tr>
</table>

Table 4. 5 Summarized feedback from managers of the university about their tree species selection and challenges for increasing tree numbers.

4.4 Discussions

Some higher education organisations have already paid attention to tree planting and maintenance on campus, and undertaken tree surveys to produce reports like "Enhancing the benefits of trees on Campus" from the University of Leeds (Gugan et al., 2019), "Tree Management Strategy" from the University of Sheffield (Winnert and Henderson, 2020) , "Tree Trail" of the University of Manchester (2021), "Tree Campus USA" (2008), "Tree Protection Standards" of the University of Kentucky (2017) and "Tree Preservation" of the University of North Texas (2009). But no previous project has wholistically assessed the terrestrial carbon stock of both, trees and soils in the greenspace of a university campus as a function of land cover to assess strategies for institutional carbon off-setting. Referencing our survey, on the campus of Newcastle University, the 0-30 cm topsoil presents a carbon storage per surface area on average 2.5 times higher than tree biomass. In terms of terrestrial carbon augmentation, introducing more trees presents the biggest opportunity as it not only adds the additional biomass carbon of trees, but also augments soil carbon according to our survey results (Table 4.1).

4.4.1 Topsoil carbon storage and mineral compositions

Across the central campus of Newcastle University, as we expected, topsoil carbon storage is statistically greater in urban woodland than lawned and the suburban sports area. This is a likely consequence of extra organic matter that includes leaves, mulches, bird droppings and the contribution of shrubs that tend to improve soil carbon content as reported previously (Edmondson et al., 2014). STC values in this study are generally higher than other reports conducted over 0-30 cm turf grass soils with tree cover in urban Melbourne (Livesley et al., 2016), urban green areas in Berlin (Richter et al., 2020), and Helsinki urban parks (Lindén et al., 2020). The soil carbon of lawn with some free-standing trees in the urban campus exceeds the values for sports ground. This could be because most sampling points in urban lawn are in closer proximity to trees (the distance ranged from 3.3 to 15.4 m, average 7.6 m), while on average the distance of soil points to the closest tree in Heaton Sports Ground is 31.6 m (nearest: 8.5 m, farthest: 75.3 m). The presence and abundance of soil microbes (Nacke et al., 2016), and soil chemical properties such as the concentration of metals (Desta et al., 2018), can be importantly influenced by the distance between soil sampling site and the tree trunk, which are all driving factors for soil carbon formation. As explained by Livesley et al. (2016), tree roots not only modify soil compaction and improve nutrient cycling, but enhance organic input to the ground, and strengthen soil carbon content. This discussion might be extended by considering land management practices and other factors, e.g., fertiliser application, frequency of grass cutting, tree ages and recreation of original soil types. Great differences in carbon storage between various land covers have previously been discussed (Pouyat et al., 2002, Lal and Augustin, 2011, Lindén et al., 2020, Richter et al., 2020). In New York City, surveyed soils reported by Pouyat et al. (2002) showed a higher organic C concentration (38%) under low density institutional land uses than in commercial land uses. The reasons resulting in this effect may be the differences in management frequency and lack of soil disturbance, or both of these.

The mineralogy of the soils is dominated by quartz, with subsidiary kaolinite and feldspar, reflecting the mineralogical composition of the geological parent material (glacial till or alluvium derived from Carboniferous sediments). SIC was reported from all soil samples in this study, and normally the source of SIC should be calcite ($CaCO_3$) (Jorat et al., 2020). However, based on diffracted patterns in Figure 4.2, calcite ($CaCO_3$) is only reported for 2 samples, reflecting the relatively high limit of detection of X-ray diffraction.

4.4.2 Trees growth for different species on the city campus

Amid a total of 67 different tree species, Large-Leaved Lime occurs more frequently (20%) than other species found in this survey, as also observed in Berlin (Richter et al., 2020). This tree is native to Europe, extending from southern Finland to the Mediterranean region, and is important as an ornamental tree, frequently seen in urban streets and parks. Within our dataset, a clear trend of larger DBH, tree height and tree cover area is shown with increasing age classifications. As the first and second largest population in this study, the tree height of mature Large-Leaved Lime (16.58 m) and Sycamore (14.73 m) is lower than the mean value from other 10 British cities (Lime: 18.1 m; Sycamore: 20.7 m) (2 sites in Wales, 6 sites in Southern England, 2 sites in Southern Scotland) (Hand et al., 2019); similarly, the tree height of mature Norway Maple (11.82 m) and Ash (13.95 m) in central Newcastle are 31% and 33%, respectively shorter than the trees growing in other British cities (Hand et al., 2019). Again, among semi-mature trees, the mean height of Large-Leaved Lime and Norway Maple summarized by Hand et al. (2019) are still 17% and 29% greater than tree parameters in this report, respectively. In some species, the tree height peaked in the early-mature age classification and remained static into the mature stage (e.g. Sycamore, Rowan), based on tree characteristics measurements (Table B. 3) (Liepiņš et al., 2016). Additionally, the observed tree canopy area for Swedish Whitebeam suggests that tree canopy could peak in the early-mature stage (Table B. 3). Generally, Norway Maple and Sycamore from mature age classification in Greater Manchester (Scholz et al., 2016) all express a variably thicker trunk (13-40%) than their counterparts in Newcastle, while fully-grown Lime and Ash from these two cities possess similar features with respect to the diameter of trunk.

For Silver Birch, on average, the trees in Newcastle are shorter than 22-year old trees in Finland, Estonia, Latvia and Russia; whereas, regarding DBH, the Newcastle Group is greater than these four Baltic groups (Viherä-Aarnio and Velling, 2017). Comparing with the study of 7,768 Silver Birch in southern Finland (Kilpeläinen et al., 2011), Silver Birch in our survey are 2-3 cm larger in diameter and 6-7 meters shorter in height. Ash is one of the most common trees across all of Europe, growing widely in mixed broadleaved stands. With respect to urban Ash in Newcastle, Latvia Ash from the 80-100 years-old group (Liepiņš et al., 2016) has an up to 14 cm smaller average DBH; conversely, relating to mean height, Ash in Latvia is almost double the height of trees in Newcastle (Liepiņš et al., 2016).

Mean environmental temperature, precipitation, sunlight time and air moisture, affected by differences in climate, are all vital factors for tree growth and the appearance of different tree dimensions (Hand et al., 2019). The reasons for the different DBH values from the same tree species between Newcastle and other European cities may lie in the tree's ability to adapt to different photoperiodic conditions caused by latitude, which in turn could explain the trunk diameter variation between Newcastle and cities in more southern regions of UK (Hand et al., 2019, Scholz et al., 2016). The decreasing tree stem height when exposed to stronger winds seems to hold true (Kronfuss and Havranek, 1999), which may thus be related to the occurrence of taller trees in other England cities compared to Newcastle (Hand et al., 2019, Scholz et al., 2016). Although wind speed was not measured in this report, a comparable dataset can be referenced (Weather Spark): no matter whether warm or cold days, average wind speed in London and Manchester all show a range of 6-15% and 5-13% weaker pattern than Newcastle, respectively. High wind speed hampers the growth of tree height: not just to escalate the risks of falling down or the loss of branches, but also increasing wind speed could cause a cooling of air, soils, leaves and meristems which are potential drawbacks during the vegetation period (Kronfuss and Havranek, 1999). It should be emphasized that tree distribution in our project is not dense because most of the trees are planted along pathways and roads where more open spaces are provided, which also means each tree is less protected by its neighbours from strong winds.

Multiple ecosystem services operated by urban trees have been positively mentioned (De Villiers et al., 2014, Hand et al., 2019, Jenkins et al., 2003, Lindén et al., 2020). For instance, street trees reduce glare reflected from the pavement, mediate a regional urban heat island, reduce air pollution, and beautify cities; conifers can form a windbreak or protect residential privacy because the needle-leaf densely grows from the bottom of the conifer stem and is evergreen; broadleaved trees lose leaves in the fall, which improves ground heat intake from the winter sun (International Society of Arboriculture, 2020). Despite diverse attractions for tree plantation, considerations related to the increase of tree numbers on campus still should be balanced with other research and business interests (Table 4. 5). Moreover, due to the function of canopy cover on rainwater infiltration and reducing run-off, the best planting density of seedlings should be carefully considered at the start of the project (Heusinkvelt, 2016, Landscape Architects- Bangkok, 2018). By figuring out the main landscape goal at the beginning of planting new trees or expanding a new green land, the time and cost consumed on the regular plant management like pruning, mulching, solving pest problems would be effectively decreased (Liepiņš et al., 2016, Limoges et al., 2018, Morani et al., 2011)..

4.4.3 The capacity of carbon capture for different tree species

The amounts of carbon stored by 490 trees calculated in this study by using allometric biomass equations is 223 tonnes. In Newcastle, urban tree carbon storages average 76.6 tonnes per hectare canopy. Carbon storges in our urban campus survey vary substantially among the eight largest number of tree species from 2.94 to 14.67 $kg{\cdot}m^{-2}$, which influences local ecosystem functions (Lal and Augustin, 2011, Nowak et al., 2013), and informs tree species selection for carbon accumulation (Burton et al., 2021, Edmondson et al., 2014, Ennos et al., 2020, Hand et al., 2019). In our analysis, some trees which are not of the main populations are likely to store larger quantities of carbon, such as the carbon storage per m^2 of tree cover from Ash and Swedish Whitebeam (Figure 4.4). This is because tree tissues (root, stem, branch, foliage, etc) of these species may have a higher carbon density (Widagdo et al., 2021), and probably these trees face a less suppressed growth condition caused by impervious surfaces (Richter et al., 2020).

Furthermore, tree age classifications play an imperative role in carbon stock outcome. The research from ourselves and Hand et al. (2019) demonstrated that carbon storage of newly planted trees is comparatively less than that of fully established trees in urban areas. For the eight largest population species modelled in our work, carbon stock increases with each successive age classification, slowly in some species (e.g. Norway Maple) but faster in others (e.g. Silver Birch) (Table 4.3**)**. Carbon stock varies not just due to different tree species, but also the climate features of sampling sites. Greater tree carbon stock values were found in other British cities (Hand et al., 2019) than the value in Newcastle, because most cities surveyed in that study are in more southerly, sunnier and warmer locations with relatively more sunlight, compared with north-eastern England, which benefits enzyme activity and provides more time for photosynthesis (Hand et al., 2019). One noted point is that the choice of allometric biomass formulas can lead to a diverse range of results when evaluating carbon storage performance in vegetation, despite inputting the same dimensions (Lal and Augustin, 2011). By using three sets of biomass equations, Vorster et al. (2020) demonstrated a substantial uncertainty of up to 75% for the estimated biomass of three tree species. Zhou et al. (2015) suggested that most biomass equations were developed based on forests, probably causing a disparity on estimating carbon stock of free-standing trees, like individual trees on the Newcastle University campus.

Tree health issues, including cavity, dead or dying branches (Boa, 2003), winter burn, fungal diseases, infestation (International Society of Arboriculture, 2020), and soil impaction (Sanders et al., 2013), importantly affect whether the trees can perform well on carbon storage. The occurrence, frequency, and out-break scale of tree health problems are combined consequences of inappropriate planting locations, wrong tree species choice, and lack of adequate planning and maintenance. For example Limoges et al. (2018) have shown that 28.54% of total tree growth condition was attributed to variables associated with street levels, geographic orientation (tree position in relation to the street), type of location, or presence of an obstruction, while 65.51% of the variation was led by tree species choice. Additionally, some urban locations are characterized as impervious and so not suitable for particular trees (Morani et al., 2011). Studying 45,500 trees across cities in the USA, Sanders et al. (2013) found significantly larger tree DBH for planting strips and non-limited soil, compared with tree pits. Conversely, the damage to infrastructure like impermeable pavements, roads and kerbs containing drainage systems caused by trees is considerable (Scholz et al., 2016), and another important consideration to take into account when selecting tree species. Land use history is another key factor influencing the appearance of trees (Heusinkvelt, 2016).

4.4.4 Carbon stock potential of urban greenspace

Many local authorities nowadays pursue a larger urban tree cover (City of Durham, Plymouth City Council, Newcastle City Council, 2019), but as scenarios 1-4 in Table B. 5-8 in

Appendix B show, the possibility of offsetting significant parts of annual carbon emissions at an institutional or city-scale is limited by the current availability of urban greenspace resources. Therefore, the involvement of rural areas through climate partnerships may become necessary to achieve net-zero targets of city institutions (Gebre and Gebremedhin, 2019). Previous work carried out in chapter 3, showed that land use change at two research farms managed by Newcastle University could make a much more substantial contribution towards offsetting institutional carbon emissions (up to 50%), than the urban campus greenspace analysed in this study (up to 1%). Similarly, city councils could seek assistance with carbon offsetting from rural partners. In return for assisting city councils with carbon abatement by planting trees or restoring peatlands, rural councils could benefit from ecosystem service payments and city council expertise to improve the rural provision of transport services and infrastructures, the upgrade of healthcare and education facilities, etc.

The limited availability of urban greenspace resources for carbon offsetting also highlights the importance and necessity of using diverse nature-based approaches (Edmondson et al., 2014, Lal and Augustin, 2011). For instance, biochar (the product of organic biomass combusted in a no or limited oxygen pyrolysis environment), as a soil amendment, potentially enhances carbon especially when using cuttings from the maintenance of urban trees or dead woods (Lal and Augustin, 2011), and we will explore this in Chapter 5. Soil inorganic carbon is around 28 % of total carbon according to the present study, while the databases of SIC over the urban lands are limited so the previous evaluations of SIC stocks from other reports may be underestimated and thus opportunities for managing SIC should be emphasized (Lorenz and Lal, 2015). Urban brownfield land, where areas have previously been used for industrial or commercial activity and become vegetated after demolition (albeit temporarily), with or without a specific design, can promote the SIC sink. Following the observed accumulation of 23 $tonnes \cdot ha^{-1} \cdot yr^{-1}$ inorganic carbon at a demolition site (Washbourne et al., 2015), an accumulation of topsoil inorganic carbon of 16 $tonnes \cdot ha^{-1} \cdot yr^{-1}$ has been reported across 20 brownfield sites in northern England, largely because of calcite precipitation, which emphasizes the importance of soil carbonation to remove CO_2 (Jorat et al., 2020). The limits to what can be achieved with nature-based carbon off-setting in urban greenspace also emphasizes the need to substantially reduce emissions when building a green city, such as switching to renewable energy systems, popularizing low carbon transport infrastructures (Newcastle City Council, 2020), deploying eco-homes (Pickerill, 2017), eco-driving and eco-charging (Ortega-Cabezas et al., 2021), and raising public sustainability awareness (Newcastle University, 2021).

4.4.5 Conclusions

Our study comes at a time when many institutions are setting ambitious targets for achieving net zero carbon. In northeast England, Newcastle University has worked together with Newcastle City Council to build the "city community forest", and university carbon managers look forward to introducing more trees on campus. This research quantified the current soil and tree carbon storage of urban greenspace on Newcastle University's city campus as a function of land-cover classifications and tree species selection to evaluate the potential of this greenspace for carbon abatement. Based on our analysis, total carbon in urban woodland soils > carbon for urban lawn soils with free-standing trees > carbon for sports grounds in suburban areas. From 490 urban trees, the eight most common tree species were divided into different tree age classifications for assessing their carbon storage potential. As a result, for tree carbon storage per m^2 of tree cover, Ash ranked the first, while Swedish Whitebeam was the best tree type in terms of carbon stock per individual tree from each species. Overall, Newcastle University could offset no more than 1.13% of its CO_2-C equivalents emissions at current rates by afforestation of its entire urban campus greenspace. Choosing the carbon value from our study to be a representative of the urban ecosystem to predict the carbon stock more widely in Newcastle City, no more than 0.95% of annual CO_2-C equivalents emissions of the city council at current rates could be offset by afforestation of its urban greenspace. This limited off-setting potential is caused by the small amount of available urban greenspace. Consequently, city institutions should first reduce their carbon emissions as much as possible before considering carbon offsetting strategies. For the hard-to-abate emissions, afforestation of urban greenspace in cities can bring many ecological and social benefits in addition to carbon offsetting, while woodland planting or peatland restoration in climate partnerships with rural councils could help city institutions achieve their net-zero carbon aims.

5. Chapter Five. Urban landscaping soil as a carbon sink through the addition of biomass and its biochar

5.1 Introduction

Globally, a total of 1,807 local governments (cities/towns) across 33 countries have declared a climate emergency by December 2020 (Cedamia, 2020) and introduced guidelines to regulate their carbon emissions for alleviating anthropogenic climate change. Currently, Canada is the leading country globally in terms of the number of local authorities (491) acknowledging the climate emergency (Shendruk, 2020). Bristol was the first council to announce the climate emergency in the UK and then another 416 local councils have followed to make similar declarations with the initiation of achieving carbon neutrality targets (Cedamia, 2020, Shendruk, 2020). In the USA, 76 local councils have passed a resolution addressing the community's impact on the climate and aiming to achieve net-zero carbon (Shendruk, 2020). Many governments have an ambitious aim to achieve city-wide carbon neutrality in the coming 10-20 years, e.g. Sydney by 2040 (City of Sydney News, 2020), Greater London by 2030 (Thorpe, 2020), and Greater Wellington Regional Council (2019) by 2030. In all anthropogenic activities, fossil fuel usage is still the dominant source for atmospheric carbon concentration increase, but complete replacement for all fossil fuel usage is unlikely to be achieved in the short term. Consequently, an approach to offset carbon emissions is inevitably required for carbon neutrality (El-Naggar et al., 2019). According to The Royal Society and Royal Academy of Engineering (2018), urban green space can offset carbon emissions through two ways: a) biomass increase of vegetation and b) carbon augmentation in urban soils. Among different carbon-offsetting methods in urban areas, biochar amendment of soils is especially attractive during construction projects, for example when establishing urban blue-green infrastructures, because landscaping works provide the ideal opportunity for biochar incorporation. If incorporated into blue-green infrastructure soil, biochar additionally offers a high potential in the passive treatment of stormwater in Sustainable Drainage Systems (SuDS) (Ulrich et al., 2015). However, previous studies have shown that soil amendment with biochar does not always simply result in a corresponding soil carbon increase (Huang et al., 2018). It may also have "priming effects" on native soil organic carbon mineralization, and affect soil inorganic carbon formation via changes in soil pH, calcium concentration, etc. (El-Naggar et al., 2019). Hence it is imperative to study the effects of biochar or alternative amendments for soil carbon augmentation under realistic field conditions.

In this broader context, Newcastle Helix (https://newcastlehelix.com/about) located in the city centre of Newcastle upon Tyne, United Kingdom, exemplifies the redevelopment of a brown-field urban space for combined commercial, recreational, and academic use. This urban redevelopment is guided by the principles of sustainability (Newcastle University, 2021) in a city that has declared a climate emergency (Bradley, 2019). According to the masterplan, about 1 hectare of the Newcastle Helix site will be covered by blue-green infrastructure designed to minimise flooding risks and reduce peak flows in sewerage systems. The site also serves as a "living laboratory" for sustainable urban development and includes the National Green Infrastructure Facility (NGIF), the UK's largest testbed for sustainable drainage systems and part of the UK Collaboratorium for Research on Infrastructure and Cities (www.ukcric.com). The NGIF integrates a range of naturally engineered systems such as green roofs and rain gardens together with a full-scale instrumented swale and experimental lysimeters. This study took advantage of these unique facilities to investigate urban soil carbon augmentation opportunities via the addition of wheat straw biomass in the form of pellets and biochar, respectively. The objectives of this chapter were: 1) to compare the impact of the two organic carbon augmentation strategies on urban soil carbon storage over a number of years; 2) to monitor effects of the amendments on urban soil hydrology; 3) to calculate the amount of carbon sequestration which could be achieved by implementing the amendment strategies for the whole green space of the Newcastle Helix site and; 4) to obtain feedback from relevant stakeholders about the practicality of the proposed carbon off-setting strategies.

5.2 Materials and methodology

5.2.1 Study site, lysimeters, and experimental design

The experimental work was conducted at the NGIF on the Newcastle Helix site (54°58'45.0" N, 1°36'49.8" W), United Kingdom. This region experiences a temperate oceanic climate with annual temperature and precipitation of 8.5 ℃ and 902 mm, respectively (MetOffice). The two lysimeters used in this study had the same volume (Figure 5. 1). Each lysimeter was 4.5 m long, 2 m wide, and 1.09 m deep. Each had an inverted pyramid-shaped bottom, which was filled with 8 mm size gravel to facilitate drainage. In each lysimeter, an 80 cm thick layer of sand (<4 mm) was placed on top of the gravel and covered with a 15 cm thick topsoil layer (8% clay, 15% silt, 40% sand, and 37% coarse fragments > 2 mm) (Figure 5. 1). The soil used here for vegetation growth and applying biochar was a landscaping topsoil from a local commercial provider, which contained fragments of construction materials. The landscaping soil texture, composition and chemical analysis can be found in Table C9-10 in **Appendix C**. With the intention of increasing soil carbon, two soil amendment materials, wheat straw pellets, and biochar produced from these wheat straw biochar pellets, were purchased from the UK Biochar Research Centre (2014), and each mixed into the topsoil layer of each lysimeter in June 2018. The wheat straw biochar was produced via pyrolysis with a residence of 15 minutes in a kiln at 700 °C **(** Table C. 2 **in Appendix C),** and was added to one lysimeter at 4.8 $Kg \cdot m^{-2}$, thereby aiming to achieve a 2% w/w TOC augmentation of the topsoil layer, which summed up to a total of 43.20 Kg of biochar (calculation steps can be found **in Appendix C**). Wheat straw pellets were applied to the other lysimeter at a higher rate aiming to achieve an equivalent carbon weight amendment as the wheat straw biochar. Thus, the amount of wheat straw pellets that was added to the lysimeter was 7.15 $Kg \cdot m^{-2}$, which summed up to a total of 64.35 Kg of wheat straw biomass pellets in that lysimeter. Pasture seeds were applied over the two lysimeters as used by the Newcastle University managed farms.

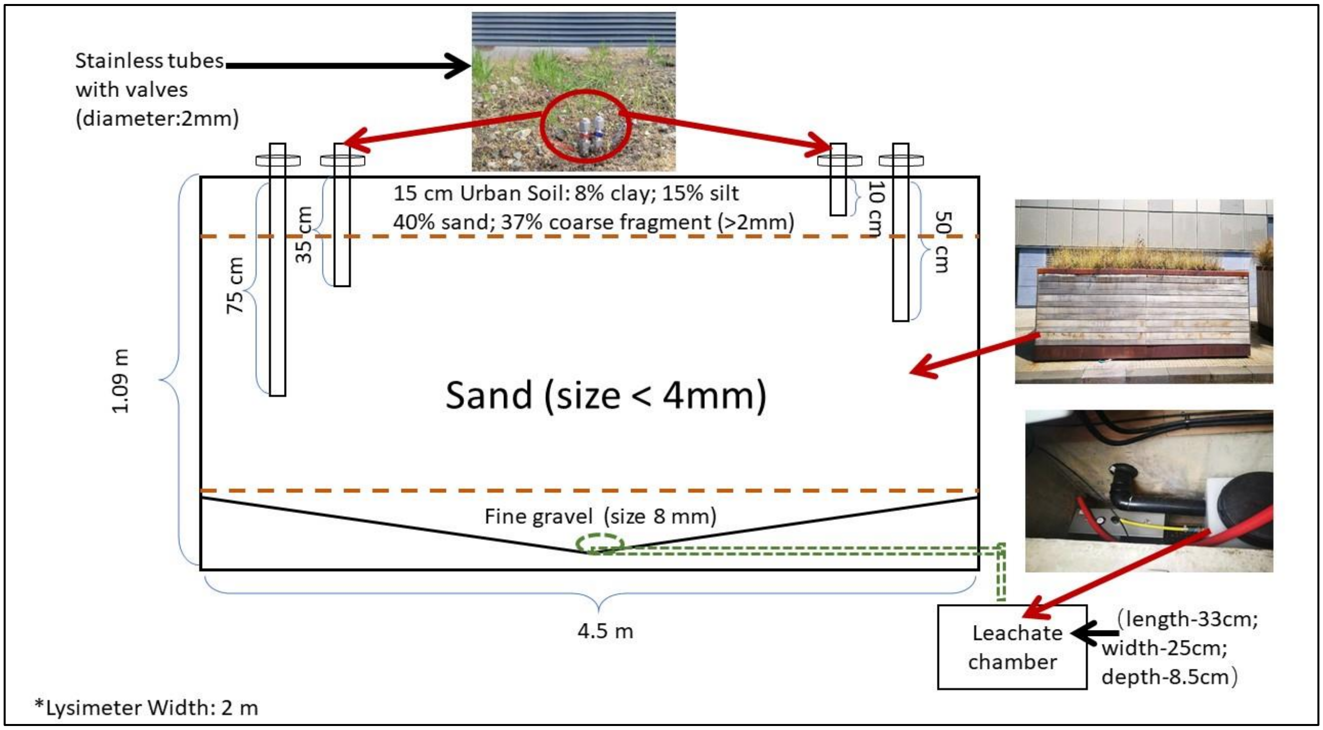

Figure 5. 1 A simple graphical description about the lysimeter.

5.2.2 Soil collection and carbon measurement

The soil carbon monitoring for this study was performed over three years from June 2018 to June 2021, with a reduced monitoring schedule from April 2020 because of the Covid-19 pandemic. Before adding the amendments to the lysimeters, samples were collected randomly from the topsoil (0-15 cm) and sand layer (40 cm). After the amendments, topsoil samples (0-15 cm) were collected with a hand auger (UMS gouge auger, 25 mm diameter) at four quadrants in each lysimeter, every three months in 2018, and every two months in 2019, plus twice a year in 2020 and 2021. Soil samples from 15-40 cm depth were collected once in 2018, three times in 2019, and twice a year in 2020 and 2021. Soil samples were prepared for % total and organic carbon analysis by a LECO-RC 612 machine (LECO Corporation, Saint Joseph, Michigan, USA), and, together with ex-situ soil bulk density, the measurements were used to obtain soil carbon density. A total of 34 lysimeter soil samples were selected for isotope analysis (20 topsoil samples; 14 subsurface samples). To understand soil hydrology, double ring infiltration tests were carried out on the surface of the lysimeters monthly (February- June 2021) (Johnson, 1963). The infiltration test was conducted in duplicate in each lysimeter on each occasion. The relevant methodologies pertaining to soil carbon measurement and calculation, carbon isotope analysis, use of carbon isotope data to establish sources of carbon in soils, and infiltration test results are provided in **Appendix C**.

5.2.3 Collection and carbon measurement of vegetation, leachate, and CO_2-C emissions

The biomass of vegetation in the two lysimeters from about 3 cm above the ground was determined by cutting in June 2019, August 2020, and June 2021 for calculating the biomass C in the vegetation. Calculation of vegetation carbon content can be seen in **Appendix C**.

A drainage pipe (Figure 5. 1) under each lysimeter collected leachate into a polypropylene chamber. One litre of leachate was collected from the chamber of each lysimeter biweekly (August 2018-February 2020) and stored in a cold room (4℃) for pH and alkalinity measurements. For determining dissolved carbon (total & organic), leachate samples were filtered with a sterile filter (0.45 um, 25 mm; VWR International, UK) to separate dissolved from particulate carbon, and afterwards the dissolved total carbon and organic carbon concentration were measured by a carbon analyser (Vario TOC cube, Elementar Analysen Systeme GmbH, Germany). Then the carbon loss from leachate was calculated by multiplying the concentration of dissolved carbon and the total rainfall (minus evaporation) between two sampling dates referenced from World Weather Online (2018-2020). More details about the calculation and leachate composition measurements are provided in **Appendix C**.

The CO_2 emissions were analysed monthly (August 2018- February 2020) by gas chromatography-mass spectrometry (Fisons 8060 GC/MD800MS). Four stainless steel tubes (2 mm diameter) in different lengths (10 cm, 35 cm, 50 cm, 75 cm) designed according to the height the lysimeters, were sealed with tube fittings and septa, and were put in each lysimeter (Figure 5. 1) so that gas samples from four soil depth layers could be obtained (Werner et al., 2004). From each tube, gases were obtained in triplicate. Meanwhile, air samples at 20 cm above the surface soil were also collected three times for measuring the CO_2 concentration in air above the lysimeters. More detailed steps in terms of gas collection and conversion of the CO_2 concentration ($g \cdot m^{-3}$) to CO_2-C fluxes ($g \cdot m^{-2} \cdot h^{-1}$) as well as the total emitted carbon mass are provided in **Appendix C**. At the end, the hourly CO_2-C flux at topsoil surface was multiplied with the total hours per month and the surface area of the lysimeter to estimate the monthly cumulative carbon lost from the topsoil layer to the atmosphere. In addition, a portable EGM-4 Gas Monitor for CO_2 (PP Systems, Amesbury, USA) was used to compare the variation of CO_2-C fluxes between two methods and its user guidance is provided as **Appendix C**.

5.2.4 Carbon mass change in the lysimeters

A carbon mass change for each lysimeter considered the amount of carbon lost via leachate and soil CO_2-C emissions to the atmosphere, and the atmospheric CO_2-C captured in the vegetation biomass and soil carbon accumulation. The percentage of each variable from the above C losses or additions to the mass of C in the amended soils could clarify the fate of C in the lysimeters.

5.2.5 Questionnaire survey of Estate and Sustainability Managers at Newcastle University

Newcastle University (2021) announced a climate emergency in April 2019 with aims to achieve net-zero CO_2 emissions by July 2030. To gather the expert opinions from Newcastle University Estate and Sustainability Managers on biochar application for soil carbon augmentation across the campus from the biochar production stage to the practical deployment stage, three questionnaires were designed. Interviews were organized with the Estate Manager of Newcastle University who manages the landscaped spaces on campus, the Director of Farms who is responsible for farms owned by the university, and the Carbon and Energy Manager. The questionnaires and interview notes are included in **Appendix C**.

5.2.6 Statistical analysis

Statistical evaluation of the data sets was performed with SPSS software (26.0, IBM Corp, Armonk, New York, USA). The comparison of carbon values between the two amended-soil systems was evaluated via a One-Way ANOVA. If additional controlling factors were involved, such as soil temperature, soil depth or water content, Post Hoc tests (Tukey's HSD) for multiple comparisons between groups were used in the statistical analysis. Plus, if one group had more than two pairs of datasets (e.g., four soil depths would have eight pairs of datasets in total), Fisher's Least Significant Difference (LSD) in Post Hoc tests was selected for comparing the pair of data within the same group. Statistical significance was considered at the $p \leq 0.05$ level.

5.3 Results

5.3.1 Soil carbon

As Figure 5. 2 presents, compared with the initial topsoil TC results obtained before the application of amendments, the topsoil TC in the last collection (June 2021) was increased by 30% in the lysimeter amended with wheat straw biochar (lysimeter BC: from 9.77±1.46 $Kg \cdot m^{-2}$ to 12.68±2 $Kg \cdot m^{-2}$), and decreased by 4% in the lysimeter amended with wheat straw biomass pellets (lysimeter WP: from 9.77±1.46 $Kg \cdot m^{-2}$ down to 9.41±1.25 $Kg \cdot m^{-2}$), respectively. Three categories of topsoil carbon in lysimeter BC were all greater than lysimeter WP, over the entire observation period, although a significant statistical relationship was absent in the inorganic carbon (One-way ANOVA, TC: $p<0.001$; TOC: $p=0.025$; TIC: $p=0.251$, Table C. 4 **in Appendix C**). The final TOC of topsoil was 40% and 9% higher in BC and WP lysimeters, respectively, compared with the initial value measured before the addition of soil amendments. Below the topsoil at around 15-40 cm depth, neither forms of soil carbon were significantly affected by the two different types of amendments (One-way ANOVA, TC: $p=0.933$; TOC: $p=0.689$; TIC: $p=0.751$, Table C. 4 **in Appendix C**). Furthermore, the variation analysis indicated that the TC in lysimeter BC, and TOC in both practices were significantly different between two soil depths (One-way ANOVA, $p<0.05$, Table C. 4 **in Appendix C**).

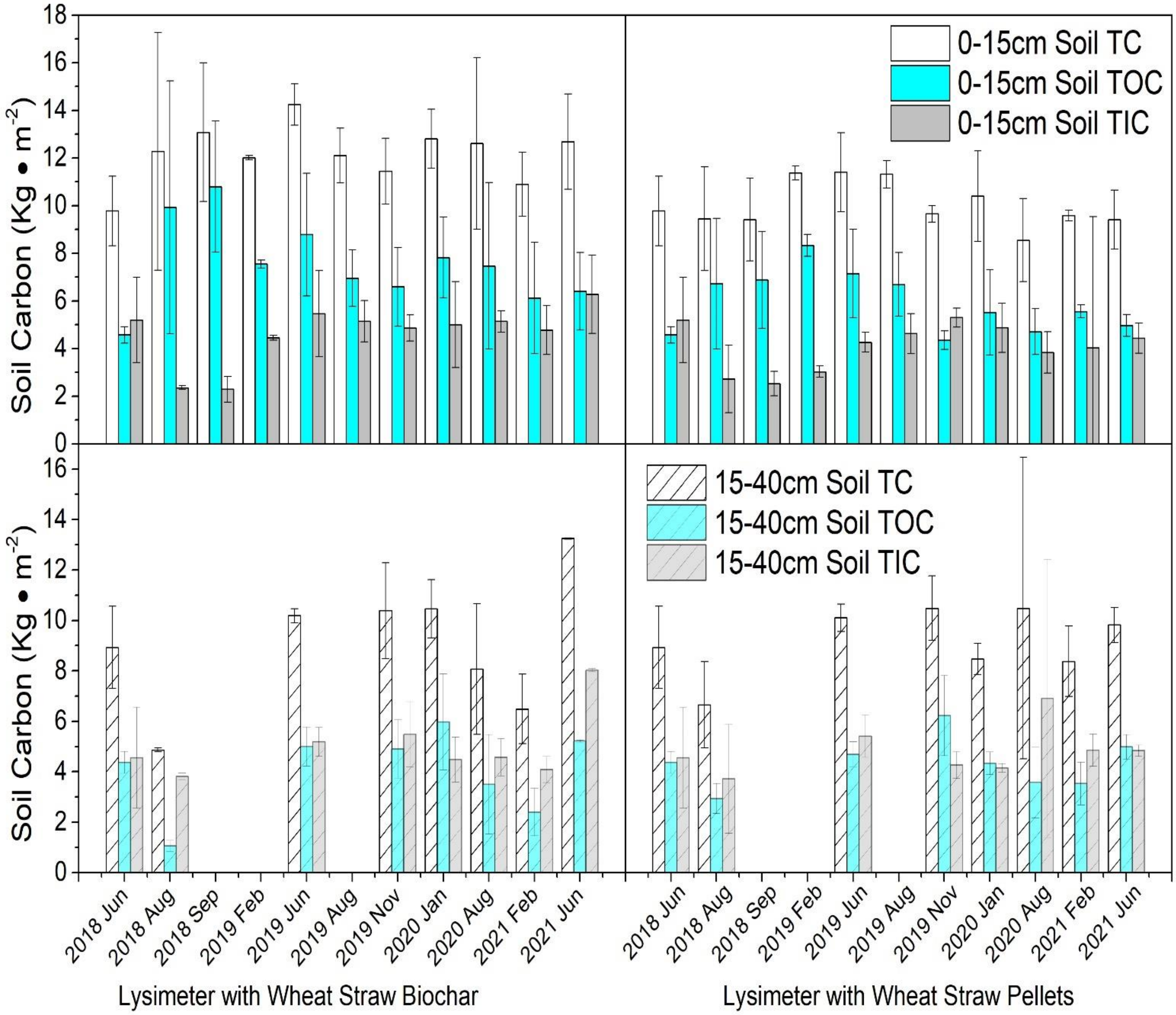

Figure 5. 2 Carbon storage (Kg·m^{-2}, Mean ± Standard deviation) of the topsoil (0-15 cm) and 15-40 cm soil in two lysimeters (June 2018-June 2021).
TC: total soil carbon; TOC: total organic carbon; TIC: total inorganic carbon.

In terms of isotopic composition, $\delta^{13}C$ in organic matter ($\delta^{13}C_{org}$) was the lowest for pure wheat straw pellets and the biochar: -28.95 ‰ and -29.53 ‰, respectively. Consequently, as Figure 5. 3 depicts, $\delta^{13}C_{org}$ values of the amended topsoil had a lower $\delta^{13}C_{org}$ throughout the whole experimental period when compared with the topsoil without treatments. The soil additives first induced a sharp decrease of $\delta^{13}C_{org}$ at both soil depths from two lysimeters while this decreasing trend in the lysimeter BC was more notable than WP. After 1 year of incubation (Spring 2019), $\delta^{13}C_{org}$ of topsoil from both environments increased steadily until the end of the experiment when the value in the lysimeter BC (-24.81 ‰) was still lower compared to WP (-24.46 ‰) (p=0.092, One-Way ANOVA, Table C. 4 **in Appendix C**), but both were much more depleted than the non-amended topsoil (-23.94 ‰). These fluctuations occurred in the 15-40 cm soil as well, but after three years the application of organic amendments led to a slightly enriched soil $\delta^{13}C_{org}$ (BC: -24.13 ‰; WP: -23.93 ‰) than the non–amended soil samples presented (-24.55 ‰) (p=0.025, One-Way ANOVA, Table C. 4 **in Appendix C**). According to Figure 5. 4, it was apparently observed that generally all $\delta^{13}C_{carb}$ and $\delta^{18}O_{carb}$ values for samples taken from the lysimeter soils corresponded to geological materials, especially limestone used in construction and as a concrete aggregate, derived from demolition waste. Overall, during the whole experiment following soil treatments, $\delta^{18}O_{carb}$ gave a more negative value in the lysimeter BC while the discrepancy in $\delta^{13}C_{carb}$ between two lysimeters was negligible, and $\delta^{13}C_{carb}$ exhibited a strong positive correlation with $\delta^{18}O_{carb}$ (p<0.001, correlation analysis, Table C. 4 **in Appendix C**). In detail, the initial $\delta^{13}C_{carb}$ and $\delta^{18}O_{carb}$ value of carbonates in the lysimeters differed between the two soil depths, being -3.25 and -9.83 ‰ respectively in the topsoil, and -4.06 and -9.03 ‰ at 15-40 cm soil. Two months after the addition of soil amendments, $\delta^{13}C_{carb}$ of topsoil in the lysimeters BC and WP was elevated to -2.45 and -1.20 ‰, respectively. After three years (June 2021), the $\delta^{13}C_{carb}$ of topsoil increased further to -1.19 ‰ in lysimeter BC, but dropped to -2.08 ‰ in lysimeter WP, respectively (p=0.804, One-Way ANOVA, Table C. 4 **in Appendix C**). Meanwhile, the $\delta^{18}O_{carb}$ value of topsoil carbonates increased from -12.45 to -9.27 ‰ in lysimeter BC, and ranged from -9.06 to -9.22 ‰ in lysimeter WP (p=0.007, One-Way ANOVA, Table C. 4 **in Appendix C**). Conversely, in the 15-40 cm layer, the $\delta^{13}C_{carb}$ showed a decrease of 0.6‰ in lysimeter BC and 3.40‰ in lysimeter WP over 36 months (p=0.706, One-Way ANOVA, Table C. 4 **in Appendix C**). The $\delta^{18}O_{carb}$ value measured from the subsurface soil profile dropped as well in both lysimeters (-5.48 to -7.68 ‰ in lysimeter BC; -3.92 to -7.97 ‰ in lysimeter WP; p=0.630, One-Way ANOVA, Table C. 4 **in Appendix C**).

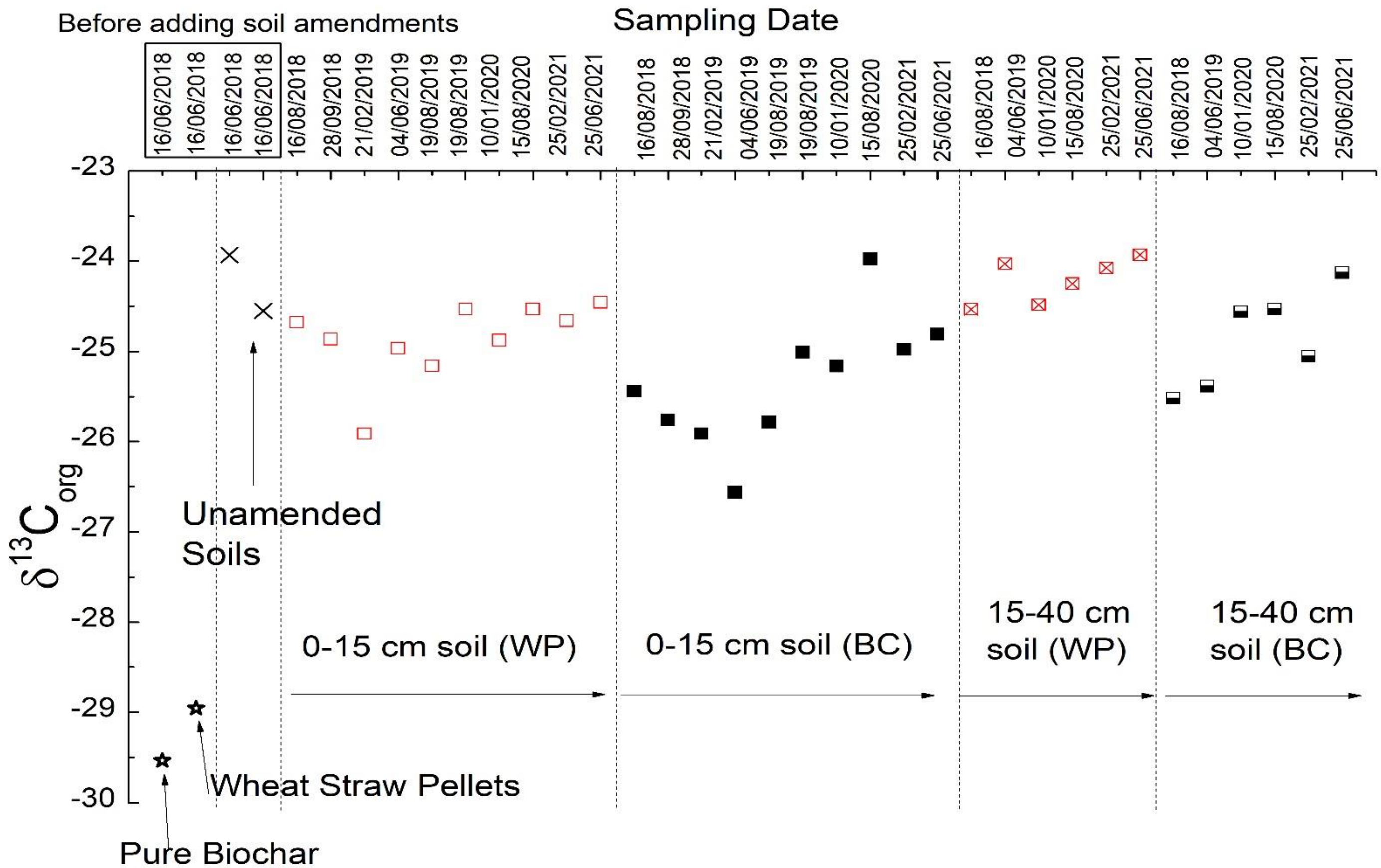

Figure 5. 3. Measured $\delta^{13}C$ for the organic matter (δ^{13} C_{org}) in the lysimeter soils.
BC: lysimeter with wheat straw biochar; WP: lysimeter with wheat straw biomass pellets.

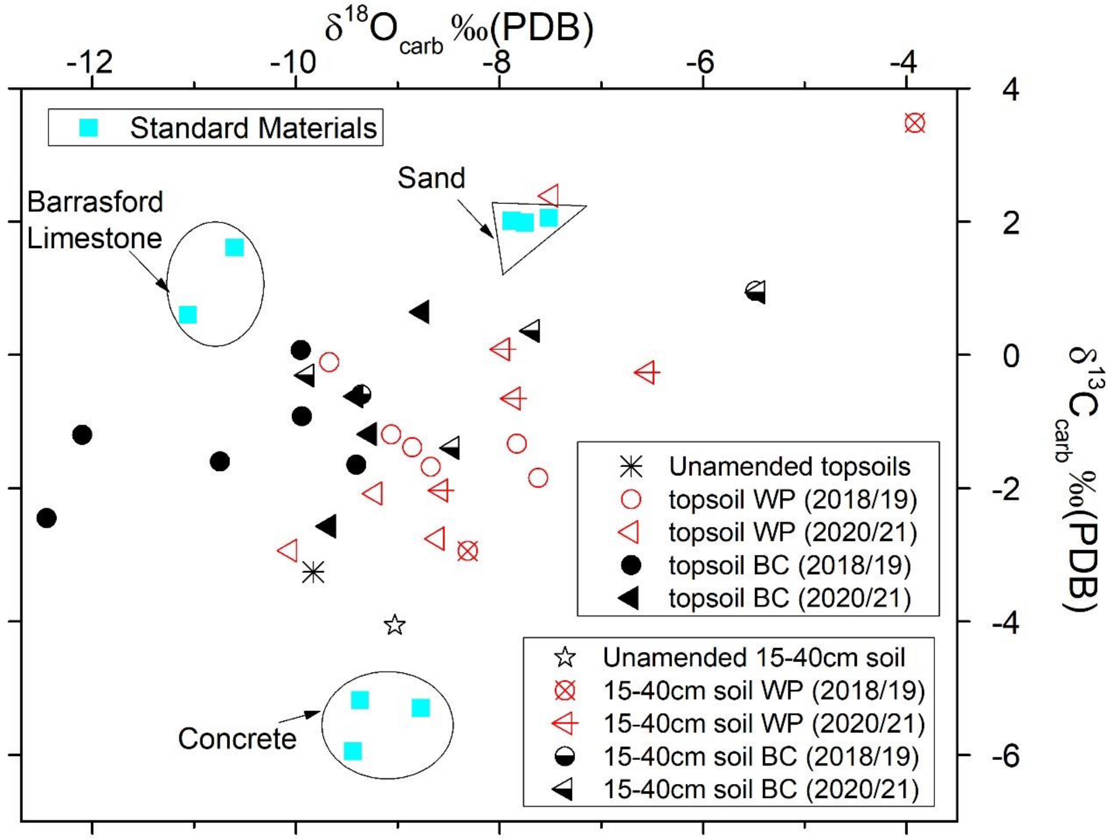

Figure 5. 4 Measured carbon isotope ($\delta^{13}C_{carb}$) and oxygen isotope ($\delta^{18}O_{carb}$) of carbonate for the materials associated with construction (sand, limestone and crushed concrete), unamended landscaping soil and amended soil samples from the two lysimeters.
BC: lysimeter with wheat straw biochar; WP: lysimeter with wheat straw biomass pellets.

5.3.2 CO_2-C flux and concentrations at four different soil layers

The trends of CO_2-C flux ($g \cdot m^{-2} \cdot h^{-1}$) from the topsoil varied greatly, no matter which measurement method was used. Thus, the related data had high experimental uncertainty and are only presented in **Appendix C** (Table C. 7). The manually installed tubes method gave an average topsoil flux value for the lysimeter BC of 0.084± 0.077 $g \cdot m^{-2} \cdot h^{-1}$, higher than WP (0.046± 0.101 $g \cdot m^{-2} \cdot h^{-1}$), but without statistically significant difference between the two data sets. The CO_2-C emission rate of topsoil in WP was initially faster than BC after two months of monitoring (Aug 2018); however, later, the CO_2-C flux in both lysimeters was highly variable. The CO_2-C emission rate derived from measurements at 10 cm soil depth recorded a peak value of 0.315±0.008 $g \cdot m^{-2} \cdot h^{-1}$ in the late spring in 2019 in BC, and peaked at 0.345±0.007 $g \cdot m^{-2} \cdot h^{-1}$ in July 2019 in WP. Additionally, the mean CO_2-C concentration recorded during the entire gas measurement period at four different soil depths in the BC lysimeter were 5-15% greater than in WP, though without statistical significance, and with great variability indicated by the error bars (Figure 5. 5; Table C. 4 **in Appendix C).** Only in lysimeter BC, across the full top 75 cm soil profile, CO_2-C concentration significantly increased with the soil depth (p<0.001, Paired samples correlations, Table C. 5 **in Appendix C).**

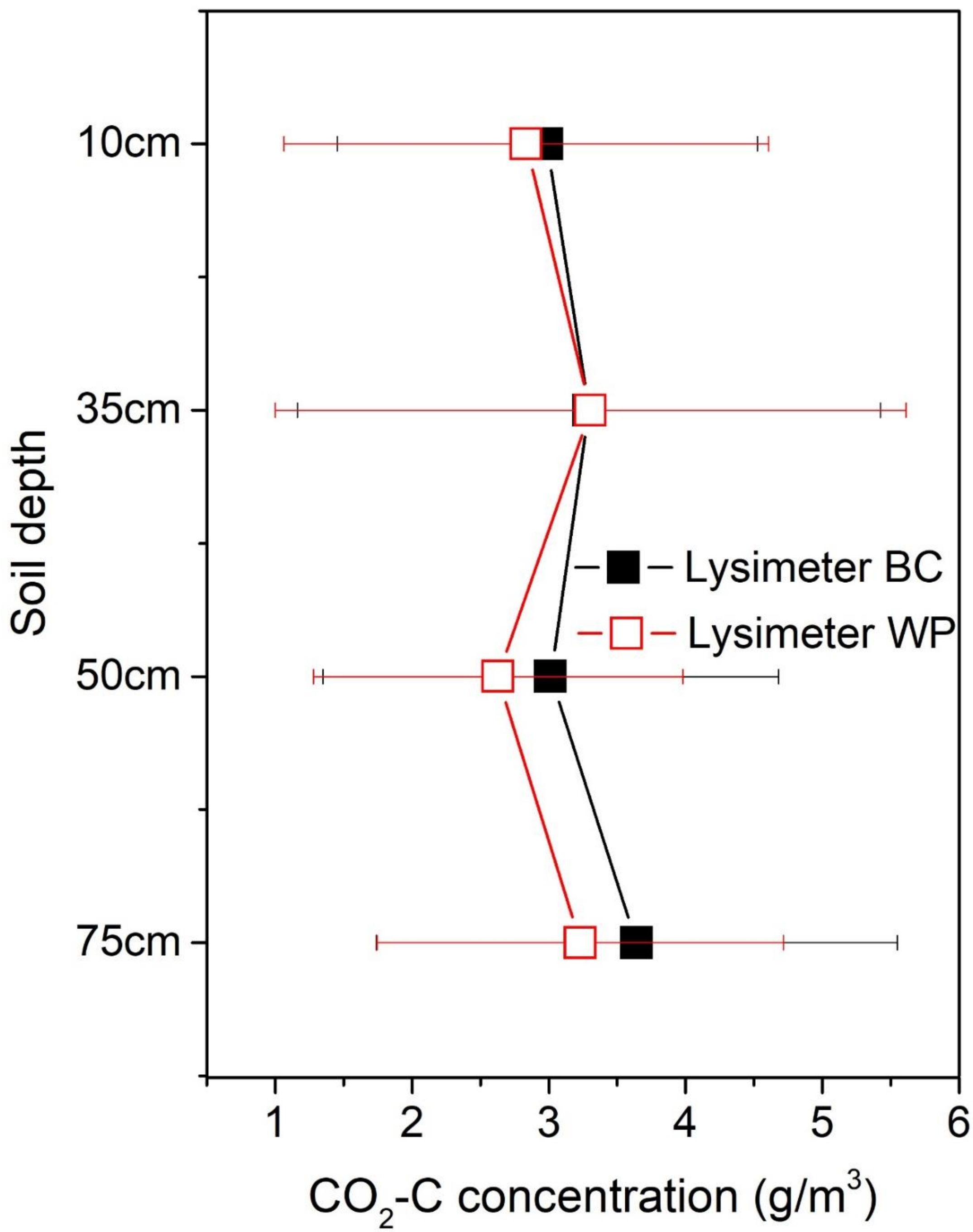

Figure 5. 5 CO_2-C concentrations ($g{\cdot}m^{-3}$, Mean ±SD) at four soil depths (10 cm, 35 cm, 50 cm, 75 cm) in two lysimeters (Aug 2018-Feb 2020).
BC: lysimeter with wheat straw biochar; WP: lysimeter with wheat straw biomass pellets. SD: Standard deviation are calculated from three replicates per treatment.

5.3.3 Dissolved C in leachates

Figure 5. 6 shows the concentration change of dissolved total carbon (DTC, $mg \cdot L^{-1}$), dissolved organic carbon (DOC, $mg \cdot L^{-1}$), pH and alkalinity of leachates over the entire observation period. The DTC concentration in leachate from both lysimeters was highest immediately following the topsoil amendment and reduced to around 100 $mg \cdot L^{-1}$ by October 2018, and then varied within the range 80-110 $mg \cdot L^{-1}$. In general, leachate from the lysimeter BC had a higher DTC in the first nine months after the amendment, but afterwards the DTC from the lysimeter WP became higher, and there was no significant difference in terms of DTC between the two lysimeters overall (One-Way ANOVA, p=0.60, Table C. 4 **in Appendix C**). On average, the DTC in leachate from lysimeters BC and WP corresponded to 93.31± 10.32 $mg \cdot L^{-1}$ and 92.13± 8.75 $mg \cdot L^{-1}$, respectively. But the mean DOC from lysimeter WP (17.90 ±5.01 $mg \cdot L^{-1}$) was significantly greater than lysimeter BC (10.49± 5.03 $mg \cdot L^{-1}$) (One-Way ANOVA, p<0.001, Table C. 4 **in Appendix C**). Additionally, pH and alkalinity in two treatments all decreased with increasing observation time. Leachate from lysimeter BC presented a higher pH and alkalinity, although the statistical correlation only existed in pH (One-Way ANOVA, p<0.001, Table C. 4 **in Appendix C**). The infiltration tests conducted monthly from February to June 2021 revealed that infiltration was highly variable, and no significant difference was found regarding to infiltration rate between the two amended soils, though the average in the lysimeter WP (2.34 cm/min) was 75 % higher than that of BC (1.34 cm/min) (Figure C. 5 **in Appendix C**).

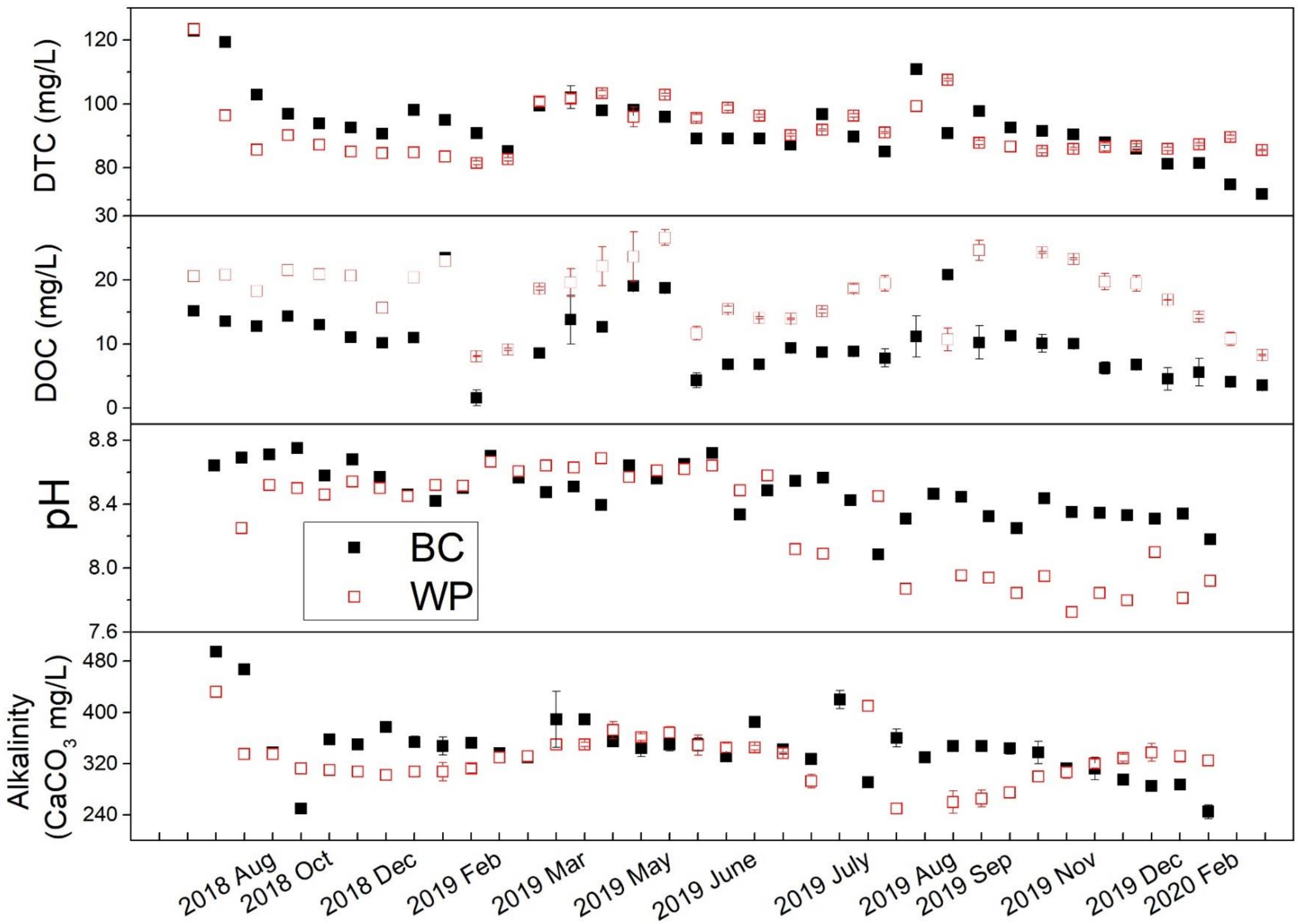

Figure 5. 6 Dissolved total carbon (DTC, mg·L^{-1}, Mean ±SD), organic carbon (DOC, mg·L^{-1}, Mean ±SD), pH, and alkalinity ($CaCO_3$, mg·L^{-1}) in the leachate from two lysimeters (Aug 2018- Feb 2020).

BC: lysimeter with wheat straw biochar; WP: lysimeter with wheat straw biochar pellets. SD refers to standard deviation of replicated measurements.

5.3.4 Carbon in clippings of above-ground vegetation biomass

Table 5. 1 presents the carbon content of the vegetative growth on the two lysimeters. In the 9 m^2 lysimeter area, the cumulative fresh wet vegetation biomass yield in the lysimeter BC accounted for 74% of the yield in the lysimeter WP in the first year, 81% in the second year, and 93% in the third year following the topsoil amendments, respectively. Over 3 years, the average carbon storage of the dried aboveground grass was 0.21 $Kg \cdot m^{-2}$ in the lysimeter BC and 0.25 $Kg \cdot m^{-2}$ in WP; but no statistical relationship presented (p=0.488, t-test, Table C. 4 **in Appendix C**).

5.3.5 Total carbon mass change at topsoil

The changes in the carbon content of the lysimeters due to various processes is displayed in Table 5. 2. Amid the various carbon change processes, the carbon loss from the lysimeters via leachate over an 18-month period accounted for 3% of the carbon added as biochar, or as biomass pellets at the initial experimental stage. For the C gain through plant photosynthesis compared to the C added as soil amendments, the ratio was similar in both lysimeters: 6% in the BC and 8% in the WP. The percentage of C loss from CO_2 gas emissions relative to the total organic C input in the two environments could not be ascertained due to high variability in the experimental measurements. However, based on a carbon mass balance it is estimated that the total amount of CO_2-C emitted from the lysimeter WP was over seven times greater than the BC. In other words, the estimated percentage of C loss caused by soil respiration to the additional C from organic input was 16% in the lysimeter BC, meaning that soil respiration and CO_2 emissions become the biggest obstacle to maintain the performance of soil carbon storage.

5.3.6 Soil water content, temperature, and bulk electrical conductivity

Figure C. 4 **in Appendix C** displays the mean of water content (cm^3/cm^3), soil temperature (°C) and bulk electrical conductivity (mS/m) from sensors placed at 10, 50 and 80 cm soil depth in the lysimeters over 3 years. At every soil depth, the water content was smaller in the lysimeter BC (p<0.001, LSD test, Table C. 5 **in Appendix C**). For instance, at 10 cm soil depth, the average water content from June 2018 to June 2021, in the lysimeter WP (23±8%) was 78% greater than that of BC (13±3%). For soil temperature, the significant difference between the lysimeters only occurred at 50 cm soil profile where the average was 2% higher in the lysimeter WP. On average the bulk electrical conductivities of lysimeter WP at each individual soil depth were all greater than the BC, though significant statistical correlation only happened around the 10 cm and 50 cm soil profiles. The mean recorded electrical conductivity of topsoil, over 36 months, was 3.92± 2.28 mS/m in the BC and 12.62± 11.42 mS/m in the WP, while both increased with soil depth **(**Table C. 5, Figure C. 4 **in Appendix C)**.

	Cutting Date	Weight of fresh clippings (Kg)	Water Content (%)	Dry mass of clippings (%)	Weight of clippings (Kg)	Carbon content of the clippings (Kg)	Above-ground biomass carbon storage (Kg·m^{-2})
Lysimeter	11/06/2019	7.0	53.0	47.0	3.31	1.57	0.17
BC	15/08/2020	10.5	49.2	50.9	5.34	2.54	0.28
	17/06/2021	7.9	60.0	40.0	3.16	1.50	0.17
Lysimeter	11/06/2019	9.5	61.5	38.5	3.68	1.75	0.19
WP	15/08/2020	13.0	51.3	48.7	6.34	3.01	0.33
	17/06/2021	8.5	50.0	50.0	4.25	2.02	0.22

Table 5. 1 The carbon content of plants in the lysimeters from annual collection of clippings.
BC: lysimeter with wheat straw biochar; WP: lysimeter with wheat straw biomass pellets. After weighing, the fresh clippings were reapplied onto the respective lysimeter surfaces, apart from a small sub-portion used for the water and carbon content determinations (no significant difference was found for the above ground biomass carbon).

	Carbon added as biochar or wheat straw (Kg)	Leaching C Loss (Kg)	Percentage of leaching C loss	C in Grass (Kg)	Percentage of grass C to the C added as amendments	ΔSoil increased C ($Kg \cdot m^{-2}$)	ΔSoil increased C (9 m^2)	Estimated C loss from CO_2 (Kg) [a]	Estimated percentage of C-CO_2 to the C added as amendments
Calculation Period		Aug 2018-Feb 2020		June in 2019/20/21		June 2018-June 2021		Aug 2018-Feb 2020	
Lysimeter BC (Average)	29.8	0.78	3%	1.87	6%	2.91	26.2	4.7	16%
SD	0.57	0.00		0.58		2.48	22.3	22.3	
Lysimeter WP (Average)	29.8	0.79	3%	2.26	8%	-0.36	-3.3	34.5	116%
SD	0.57	0.00		0.66		1.92	17.3	17.3	

Table 5. 2 The total carbon mass change (Kg, Mean ± SD) of topsoil layer in the two lysimeters.
BC: Lysimeter with wheat straw biochar; WP: Lysimeter with wheat straw pellets.
[a]: Calculated as the estimated C loss as CO_2-C (Kg) to the atmosphere is equal to the C addition as biochar or wheat straw pellets (Kg) minus the ΔSoil C (kg) increase between the final amended (year 3) and the unamended soils (year 0) plus the C addition in the form of vegetation biomass (Kg) minus the C loss with leachate (Kg). Therefore, the estimated C loss from CO_2 can be obtained by this carbon mass balance equation.

5.4 Discussion

5.4.1 Effects of the soil amendments on soil carbon

Our experiments showed that amendment of an urban topsoil with wheat straw biochar pellets lastingly improved soil carbon after three years, while addition of the equivalent amount of carbon in the form of the original wheat straw biomass could not increase soil carbon lastingly (Figure 5. 2). These findings are in line with reports for agricultural soils (Carmo et al., 2016, Huang et al., 2018). With a biochar application rate equivalent to 48 tonnes·ha^{-1}, the TC, TOC and TIC of the urban topsoil in the present study was enhanced by 30%, 40% and 21%, respectively, 3 years after the biochar application. Similarly, previous studies have shown that soil amendment with biochar is able to enhance soil carbon contents (Carmo et al., 2016, Paetsch, 2018, Somerville et al., 2020), e.g. at the end of six years, soil carbon increased from 17% to 41% in forest soil for biochar application rates of 2.5 and 25 tonnes·ha^{-1} across the western USA (Sarauer et al., 2019). Across the arable lands, however, there was a 3.17% increase of TC in plant biochar-treated soil after 2 years with an application rate of 11.5 tonnes·ha^{-1}, compared with untreated soil (Saletnik et al., 2018). In our experience, the increase of topsoil TOC in the lysimeter BC (40%) is significantly greater than the lysimeter WP (9%). By contrast, in the 0-20 cm soil layer, Yang et al. (2017) have observed a higher increase of TOC in the maize stover application (25%) compared with a maize-stover biochar treatment (19%) after three years. Higher carbon in the topsoil of the lysimeter BC confirms the hypothesis that pyrolysis of wheat straw biomass reduces its biodegradability, and hence C mineralization in the soil environment (Zhu et al., 2017). This demonstration is in accordance with the study of Song et al. (2019) who cultivated crop soils using cotton straw biochar. No statistical difference in C contents in the subsurface soil was found between the two lysimeters, which indicates that the wheat straw biochar amendment was retained in the topsoil layer.

5.4.2 Effects of the soil amendments on soil carbon isotope data

Carbon isotope labelling could trace the origin of carbon change in soil organic matter and pedogenic carbonate, and further calculate the carbon contribution from parent materials with turnover rates, e.g. dividing the soil respiration components caused by autotrophic or heterotrophic metabolism and distinguishing CO_2 flux from terrestrial or oceanic ecosystems (Wang et al., 2015, Dong et al., 2019). The key to interpreting the carbon isotope data is the difference in $\delta^{13}C$, which reflects fractionation of ^{13}C and ^{12}C. $\delta^{13}C$ for C3 and C4 plants is different, as a consequence of differing discrimination against ^{13}C for the two dominant photosynthetic pathways. The $\delta^{13}C$ of C3 plants, like wheat, is around -27 ‰, which is similar to our data (wheat straw: -28.95‰; wheat straw biochar: -29.53 ‰), whereas $\delta^{13}C$ of C4 plants (e.g. maize, sugarcane) is more enriched (-13 ‰) (Christensen et al., 2011, Wang et al., 2015). In addition, $\delta^{13}C$ from atmospheric CO_2 and pedogenic carbonate is less depleted. Considering the soil carbon change is driven by root respiration, oxidation of soil organic matter and decomposition of surface litter, the $\delta^{13}C$ from carbon input sources such as vegetation and litter alters the $\delta^{13}C$ in the soils during cultivation (Christensen et al., 2011). Also, the variation of chemical and biological stability of biochar would occur over the incubation time because of weathering. The active carbon pool from biochar might become less prominent with the observed time, while more stable C is retained in the biochar, all of which drive the different changes of $\delta^{13}C$ from biochar and amended soils (Naisse et al., 2015, Zhang et al., 2022). In terms of $\delta^{13}C$ of soil organic carbon, we hypothesized that $\delta^{13}C_{org}$ in soils decreased with the increase of soil organic carbon because both wheat straw residues and its biochar present a lower $\delta^{13}C_{org}$ (positive priming effect) (Christensen et al., 2011, Menichetti et al., 2014, Song et al., 2019, Wang et al., 2015). In both lysimeters, $\delta^{13}C_{org}$ values for soil samples from the entire observation period were all lower than the unamended landscaping soil collected before treatment, which is in line with the hypothesis (Song et al., 2019). Similarly, a 180-days experiment, applying cotton straw and its biochar into cropland soils, has observed a more depleted $\delta^{13}C_{org}$ in cotton straw biochar amended soils, which indicates the bonding of carbon derived from organic matter in the soil as more liable organic matter is oxidized (Song et al., 2019). In the present study, a more depleted $\delta^{13}C_{org}$ can be always found in lysimeter BC (One-way ANOVA, p=0.092, Table C. 4 **in Appendix C)**, which illustrates that the organic matter from the amendment is better preserved in the soil as BC, compared to the WP amendment. The $\delta^{13}C_{org}$ values for both soil layers from the two lysimeters decreased initially following the amendments and one year later increased back in the direction of the signal at the start of experiment with minor fluctuations, as expected. Nevertheless, a significant difference of $\delta^{13}C_{org}$ between two lysimeters existed in subsurface soils alone. Following the initial drop, the $\delta^{13}C_{org}$ in the lysimeter BC returned back more slowly than in the lysimeter WP, showing the greater stability of the BC amendment. All these trends are in accordance with the variation of TOC which shows an enrichment with the amendments and then a decrease over time. This relationship is consistent with other long-term bare fallow studies in five European cities reported by Menichetti et al. (2014) who thought the decrease of TOC resulted via two losses of carbon from soil: either being the gas emitted or being the dissolved carbon in the leachate. As a result, the experimental duration seems to play a role in the performance of carbon sequestration in the soil amendments as organic inputs from additives will cease at one time point, as well as their potential priming impacts on depressing the microbial activity and metabolism (Menichetti et al., 2014).

The carbon isotope analysis involving the carbonate can help recognize how soil inorganic carbon changes. In comparison to the engineered soil collected before treatment, $\delta^{13}C_{carb}$ was markedly more enriched in the topsoil of both lysimeters, but this shift was less pronounced in the WP as compared to the BC lysimeter. The low $\delta^{13}C_{carb}$ of the landscaping soil could be due to its content of some demolition wastes such as concrete, which has a very low $\delta^{13}C_{carb}$ of -18 to -32‰ (Jorat et al., 2020), or fluctuates around -5.5 ‰ as for Technosol soils in a separate experiment at Cockle Park Farm of Newcastle University (Figure 5. 4) (Son et al., 2021). From the TIC measurements (Figure 5. 2), there was initially a notable reduction in the soil inorganic carbon content of the topsoil in both lysimeters. Such a large loss of CO_2 from carbonate dissolution may result from the stimulated activity of bacteria following soil amendment (Sheng et al., 2016). In this study, the soil TIC was then gradually replenished over time, and more so in the BC as compared to the WP lysimeter. According to the $\delta^{13}C_{carb}$ signals, the newly formed soil TIC had a more enriched $\delta^{13}C_{carb}$ values than the inorganic carbon initially present in the landscaping soil, a portion of which was lost following the amendments. Less formation of fresh soil inorganic carbon over time may then explain why the $\delta^{13}C_{carb}$ in the WP lysimeter remained below that of the BC lysimeter. Furthermore, some of the organic carbon from the amendments with a low $\delta^{13}C_{org}$ may have been mineralised and become part of the newly formed carbonates. This probably occurs to a greater extent for the less stable wheat straw pellet C_{org}, also resulting in a lesser $\delta^{13}C_{carb}$ signal for the topsoil in the WP as compared to the BC lysimeter. Greater TIC formation in the BC lysimeter would be facilitated by 1) the supplement of Ca^{2+} from ash content in biochar (Dong et al., 2019); 2) a higher pH of the biochar amended soil environment; 3) low electrical conductivity; 4) the decreased soil water content (Khormali et al., 2020). The latter two factors have been discussed by Khormali et al. (2020) who surveyed six various soil profiles over nine different locations in northern Iran, containing four soil taxonomic types, where all sampling points present their identical geological feature in terms of latitude, mean annual precipitation and air temperature. Decreasing electrical conductivity and soil moisture, representing a less arid environment, may favour the accumulation of carbonate (Khormali et al., 2020). This suggestion is in line with our measurements that at 10 cm soil depth, soil electrical conductivity and water content of lysimeter WP is importantly five times, and two times greater than BC, respectively. Khormali et al. (2020) discussed the impact of rainfall and air temperature on carbonate formation, however, these two parameters were the same in both lysimeters in this study. Infiltration rate is likely also relevant for the dissolution-formation of soil carbonate (Huth et al., 2019), especially over periods with large rainfall events, which significantly links the movement of Ca^{2+}, the partial CO_2 pressure in the soil. The average infiltration rate for the wheat straw biomass-amended topsoil in the present study is 75% faster than for BC amended topsoil. Although lacking a significant correlation, the observed difference could be an explanation of enhanced weathering of carbonate leading to the loss of soil inorganic carbon in the lysimeter WP (Figure 5. 2).

5.4.3 Effects of the soil amendments on C loss via leachate

Several studies have stated that adding biochar into soil might be an approach to reduce the leaching of carbon and nutrients (Iqbal et al., 2015, Liu et al., 2016). As we anticipated, the dissolved organic carbon (DOC) concentration was significantly lower in the lysimeter BC (Zhu et al., 2017). The DOC change in soils is attributed to the partial decomposition of soil organic matter into leachable fractions such as fulvic acids, while dissolved inorganic carbon (DIC) is driven by the equilibrium of bicarbonate, carbonate, and $CaCO_3$ (Duffy and Nikolaidis, 2015). On the one hand, a significantly lower DOC in the lysimeter BC might be the result of strong absorption capacity of biochar for organic molecules due to its surface area in the pore space (Zhu et al., 2017); on the other hand, a higher DOC found in the lysimeter WP could reflect the wheat straw pellets being a more easily decomposable substrate than the biochar, and in the longer term also the enhanced vegetation yield of the WP lysimeter. Our three years of data substantiate the short-term trends observed in a 35-days project in which sandy loam soil was amended by maize straw or its biochar (Zhu et al., 2017). Zhu et al. (2017) firstly ascribed the greater DOC in leachate of straw treatment to the higher decomposition of straw, and secondly stated that less DOC leaching in the biochar treatment could be accounted for the enhanced sorption on biochar pore surfaces. In contrast, the DOC leaching from compost mixed with forest slash biochar in the report of Iqbal et al. (2015) was higher than the compost-only system, but without statistical significance. Iqbal et al. (2015) believed the low sorption of biochar in their study was caused by fast flows, so contact time is insufficient to achieve an effective sorption. Similar results were observed by Liu et al. (2016), who claimed that liable biochar carbon caused a significant higher DOC in soils added with the mixed crop straw biochar. In the present study, DOC was a lesser contributor to the DTC in leachate than DIC. For DTC, amendment effects were less clear cut, with greater DTC leaching initially from the BC as compared to the WP lysimeter, and an inversion of this trend towards the end of the observations. Similar observations for alkalinity corroborate these findings, since alkalinity typically reflects the DIC content of leachate (i.e.

carbonate and biocarbonate). Overall, C loss via leachates over the observation was only about 3 percent of the C added to the topsoil with either type of amendment.

5.4.4 Effects of the soil amendments on above ground vegetation biomass

In agriculture, applying biomass residues or biochar to soil is a feasible option for improving crop productivity (Huang et al., 2018, Lorenz and Lal, 2014). We observed a higher vegetation productivity in soils incorporated with wheat straw in pelletized form, as compared to wheat straw biochar, during three consecutive years **(**Table 5. 1). According to Carmo et al. (2016), the increase in crop productivity was sensitively and positively affected by soil electrical conductivity, being in line with our findings **(**Figure 5. 6 **in Appendix C**). An equal amount of organic C was initially embedded in the two lysimeters, while the total amount of additives was different, and thus the proportions of salts and ions vary initially. The higher amount of wheat straw pellets brings more nutrients, which may explain the greater dried grass biomass obtained in the lysimeter WP (Table 5. 1) (Carmo et al., 2016). Similarly, this happened in rape yield by applying rape straw and its biochar, respectively (Huang et al., 2018). High pyrolysis temperature has a strong influence on deceasing nitrogen content of feedstock materials (Chatterjee et al., 2020), and thus the partial loss of nitrogen in the biochar production, and hence reduced soil fertility, may be the reason for the less vegetation growth (Table 5. 1). On the other hand, Scharenbroch et al. (2013) found mixed pine biochar increased tree biomass of Sugar Maple and Honey Locust, while wood chip treatment did not affect tree growth. Saletnik et al. (2018) acknowledged that the increased concentration of macro-microelements of biochar, e.g. phosphorus, potassium, magnesium, due to the fast exchangeable ion sorption, could contribute the extra crop growth. Lorenz and Lal (2014), however, hold an opposite view that chemically active surfaces in biochar stimulate the mobility of soil nutrients, instead of improving the cation exchange capacity (Manyà, 2012).

5.4.5 Effects of the soil amendments on soil-atmosphere CO_2 emissions

In our research, CO_2-C fluxes from topsoil in the both lysimeters were calculated from monthly measurements and were highly variable, ranging from 0.001 to 0.35 $g \cdot m^{-2} \cdot h^{-1}$ when determined by the manually installed tubes method, and from -0.002 to 0.13 $g \cdot m^{-2} \cdot h^{-1}$ according to the portable CO_2 analyzer (Table C. 7). Consequently, for the measured CO_2-C fluxes, the absence of statistically significant differences among biochar amended and no-biochar systems might be first contributed to insufficient resolution in the data in both space and time (Sarauer et al., 2019). A more apparent relationship between different amendments practices with an important difference tends to happen in short-term research (Ameloot et al., 2013, Wang et al., 2014), but for the longer-term research presented in this study, more regular sampling across multiple locations on the surface of each lysimeter would have exceeded the project's financial and human resources. A mass balance approach was therefore used to assess the overall carbon losses which had occurred as CO_2-C from each lysimeter to the atmosphere over the three-years observation period. Consequently, the total estimated CO_2-C loss over 18 months caused by soil respiration is 4.7 Kg in lysimeter BC and 34.5 Kg in WP, respectively. This outcome is anticipated because biochar is assumed to prevent carbon emission due to its lower content of labile C (Yang et al., 2017).

5.4.6 Carbon sequestration opportunity during urban site redevelopments

The redevelopment of urban areas as part of green city planning offers opportunities to policy-makers, scientists and practitioners to consider urban soil carbon augmentation as an off-setting measure and contribution towards net-zero emission targets (Edmondson et al., 2015, Rawlins et al., 2008, Renforth et al., 2011, Scharenbroch et al., 2013). If the present study considers the soil carbon augmentation of the BC over the WP treatment obtained by the end of the three-year period as representative for the longer-term carbon sequestration benefit of applying pyrolyzed instead of unaltered biomass to soil, then the wheat straw biochar application in urban soils could capture carbon around 2.91 ± 2.48 $Kg\cdot m^{-2}$. Urban development represents a substantial 8% of land use in UK (1.8×10^{6} hectares) (Office for National Statistics, 2019), and urban land development typically involves earth works which then provide an opportunity for embedding amendments. If such an amendment discussed above are applied to the entire green space (1 ha) in the Newcastle Helix site master plan, it would store a total of 30± 25 tonnes of carbon. While this estimation may seem to be a large amount, it is minor compared to the total stage 1 and 2 CO_2 equivalent -C emissions of Newcastle University in 2019/20 (6,406 tonnes of CO_2 equivalent -C) (Newcastle University, 2021). Therefore, this calculation shows that such amendments would need to be used at a very large scale to be relevant for off-setting urban carbon emissions. However, in the context of blue-green infrastructure, additional benefits might strengthen the case for soil biochar amendments. Several studies suggest that biochar can facilitate the water cleansing process in stormwater management systems (Borah, 2020, Karhu et al., 2011, Renforth et al., 2011, Scharenbroch et al., 2013). Borah (2020) has summarized potential benefits of using biochar in the context of urban stormwater management, including contaminant removal (Scharenbroch et al., 2013). For example, biochar can help ameliorate *Escherichia coli* loads from stormwater owing to its highly porous surface area as well as a strong attachment with hydrophobic attraction to bacteria (Mohanty et al., 2014).

5.4.7 Stakeholder perspectives

Interviews with key stakeholders revealed that several challenges exist in implementing a soil amendment strategy using biochar produced from local crop residues, from the selection of biochar sources to biochar production and application (Table 5. 3). Firstly, insufficient crop residues are left at the university-owned farms after use on site and crop residues are more likely to be left in the agricultural land to recycle the nutrients into the soil, thus utilising farm residues for producing biochar is not attractive from a conventional farm management perspective (Haefele et al., 2011). On the other hand, the Estate Manager of Newcastle University (2021) would be willing to use biochar due to its desirable function of carbon storage if the technique could benefit the institutional target to achieve net-zero carbon emissions by 2030. Even though this small-scale trial was successful, estates management would still be concerned about the problems in terms of dust, storage, haulage, and other factors related to the practical feasibility of biochar application. Finally, the Sustainability Team had questions about the embodied carbon of biochar, the final appearance of soil following biochar application and how often to repeat the biochar application (Table 5. 3).

Objectives	Estate Manager	Farm Director	Carbon& Energy Manager
Key response			
1. The responsibility of department in achieving "Carbon Management Plan"	➢ Using more electric vehicles and battery-powered machines and less petrol and diesel ➢ Selecting the grass seeds which performs well in carbon sequestration ➢ Assessing carbon footprint of grasslands ➢ Building a carbon capture garden ➢ Planting more trees	➢ Develop the farm-scale "Carbon Strategy" to supplement the university's carbon plan ➢ Calculating carbon sequestered in farms by carbon calculators ➢ Improving the pervious projects related for farms development ➢ Setting up the effective working system ➢ Establishing carbon measurement metrics ➢ Altering the cultivation system ➢ Assessing the costs farm spend ➢ Monitoring soil productivity	➢ Planting more trees to enhance the biodiversity ➢ Balancing the act between carbon management and other business activities. i.e. purchased goods and services, travel ➢ Engagement - promote sustainability and climate action within and outside of our institution, such as working with farms about planting trees and exploring renewable energy ➢ Monitoring energy consumption and checking any faulty system control or operation as well as improving the energy performance of our estate and increasing our renewable energy generation. ➢ Capital goods – minimise embodied carbon associated with our construction activities. ➢ Investments – minimise the carbon footprint associated with our endowment investment portfolio. ➢ Circular economy – embed waste hierarchy principles and decrease our total waste mass. ➢ Research and education – expand our sustainability curriculum and create our 'Living Lab'. ➢ Leadership and governance – demonstrate leadership in the fight against the Climate Crisis, and support others to do so.

2. Concerns for biochar production and application	➢ Parent material of biochar ➢ 100% Safe? ➢ Storage requirements ➢ Safety issues from dust ➢ Leaching to drainage system ➢ Large quantity needed? ➢ The way to apply biochar is convenient or not	➢ Substantial consumption in electricity ➢ The lack of subsidy	➢ Biochar impacts on vegetations ➢ Sources to purchase biochar ➢ Transportation of biochar
3. Challenges facing for biochar popularization	➢ Constraints of available space around city centre ➢ Biochar stability	➢ For crop residues-derived biochar, insufficient crop residues left ➢ For timber biochar, the specific area of farm fields is not permitted to convert to woodlands. ➢ The feasibility of experimental outcome from a small-scale trial	➢ The endpoint of biochar in soils ➢ The embodied carbon of biochar ➢ Recycling of biochar ➢ The frequency to spread biochar

Table 5. 3 Bullet points of key responses from executives of Newcastle University about biochar production and application, and furthermore its function on carbon sequestration (Full interview response can be found in **Appendix D**).

5.5 Conclusions

Urban green space plays a vital role in providing an environmentally friendly and harmonic living area for citizens and can contribute to mitigating climate change by increasing carbon stocks in a limited space, whilst also reducing climate change impacts such as flooding risk with the implementation of Sustainable Drainage Systems. The project in this chapter applied wheat straw biomass pellets and its biochar in a landscaping soil to compare the respective amendment benefits for soil carbon over three years. It was found that the carbon accumulation in the biochar-amended topsoil is 2.91±2.48 $Kg \cdot m^{-2}$ at the end of the observation period, while carbon loss happened in the biomass pellet-amended topsoil (0.36±1.92 $Kg \cdot m^{-2}$). According to the carbon storage potential calculated in this study, the overall green space (1 ha) of a current city centre redevelopment site in Newcastle upon Tyne, Northeast England, would have the ability to accomplish carbon sequestration of around 30 tonnes under wheat straw biochar application. The introduction of biochar from crop residue biomass into urban landscaping may be an efficient opportunity to increase terrestrial carbon, as well as a means for strengthening the ecological credentials of green city designs and urban stormwater management. However, the limited availability of urban land for the implementation of carbon storage measures is a major challenge when trying to off-set institutional or even city-scale carbon emissions. Furthermore, the availability of crop residues for biochar production, transportation and implementation issues were concerns voiced by potential stakeholders in such off-setting schemes. These insights emphasize the need for institutions to first drastically reduce their current carbon emissions, which would then make the off-setting of hard-to-remove, residual emissions a more realistic prospect as these institutions strive to achieve their net-zero carbon emission aims.

6. Chapter Six. General discussions and conclusions

6.1 Implication of the key findings for terrestrial carbon management by institutions

This thesis explored the links between different land managements and their corresponding terrestrial carbon storage, which involved soil sampling at farms and urban areas, tree dimensional data collection, and a three-year lysimeter study investigating the influence of wheat straw and its biochar on urban landscaping soil carbon sequestration, respectively. These works determined for the first time the total terrestrial carbon stock of Newcastle University (2021) which ambitiously plans to achieve net-zero carbon by 2030. Actions by universities to set and pursue ambitious targets to reduce carbon emissions can set an example for other organizations' efforts to start reducing institutional emissions. Typically, universities possess a variety of academic resources and cooperation opportunities to develop sustainable schemes. Previous highlights regarding the carbon abatement strategy, led by the sustainability team of Newcastle University, have included the installation of ultra-low/zero-carbon technologies, establishing energy policies, circular economy, auditing the sustainability of taught courses and upgrading evaluations for procurement, etc (Newcastle University, 2021). However, in line with other universities across the globe, Newcastle University's carbon abatement strategy did not quantitatively consider management of terrestrial carbon in its estate, i.e., the carbon stock of soil and trees of its estate, which comprises the urban campus, peri-urban sports grounds and two rural research farms. A lack of available data limits the carbon managers' understanding of the importance of land management and the potential for carbon accumulation in the soils and trees by land use conversions to off-set institutional carbon emissions.

As a result of this thesis, the quantity of terrestrial carbon pool in land managed by Newcastle University, across 25 ha of urban greenspace and 803 ha of suburban farm fields, is now known to be around 107, 988 tonnes, being 17 times the current annual carbon emissions (6,406 tonnes of CO_2-equivalent C) for Newcastle University. For these terrestrial carbon stocks, 96% were contributed from farms and another 4% resulted from central campus lands.

Since the current terrestrial C pool of Newcastle University has now been established, the work presented in this thesis could also explore opportunities to enhance this pool as a way of off-setting institutional carbon emissions. The key conclusions that arose from this research about such institutional opportunities are as follows.

6.2 Effects of land management over the farms and urban greenspace on carbon sequestration

Location and soil depths significantly affected the terrestrial carbon stock, and soil carbon storage in NF (Nafferton Farm) was overall higher than CPF (Cockle Park Farm), in either soils sampled from arable land and permanent grassland, or under coniferous and broadleaved cover, likely reflecting differences in local geology and soil forming processes. At each farm, woodland showed a greater capability of sequestering soil carbon than agricultural land. This discrepancy was larger in CPF. The soil carbon accumulation from permanent grassland at CPF was statistically 18% greater than that of arable land at CPF. At NF, the soil classifications between conventional and organic did not have a significant influence on soil carbon storage. Across the woodlands at two farms, the division according to leaf types did not significantly influence the soil C sink, although soil carbon storage at the coniferous woodland was generally higher than for broadleaved. Additionally, the different planting time at CPF woodland did not statistically affect soil carbon accumulation but the larger soil carbon concentrations were measured at the long-established woodland. At the same time, across the central campus of Newcastle University, located in the core of Newcastle City, the divisions of soil categories (woodland soils, soils in sports ground and soil in the lawns with free-standing trees), importantly affected the 0-30 cm soil carbon storage. Following trends at the farms, across the Newcastle University urban greenspace, soil carbon contents in the lawns with sparse tree cover or the sports ground were significantly smaller than that of urban woodland. Therefore, combining all soil carbon results from urban green lands and suburban farms, converting the lawns or agricultural land to woodland, or converting arable land to grassland rather than just altering crop cover, would be the most efficient way to enhance soil carbon, with the additional benefit of carbon storage in the tree biomass. Coniferous trees at CPF showed a higher potential for sequestering carbon in biomass than broadleaved; at the same time in the central campus, Ash exhibited the greatest ability of carbon sequestration in terms of carbon storage per square meter of tree cover, however, Swedish Whitebeam was the most promising tree type among 67 tree species when valuing its carbon accumulation as an individual body.

6.3 Effects of biochar on carbon sequestration and leachate quality in landscaping soils

In a 2 lysimeters study over three years where the 0-15 cm top layers of landscaping soils were amended with 2% w/w organic carbon (one with wheat straw biochar, the other with wheat straw pellets), the biochar addition produced an important soil carbon increase of 2.91 ± 2.48 $Kg\cdot m^{-2}$, as opposed to the raw crop residues treatment which did not augment soil carbon in the longer term. Meanwhile the $\delta^{13}C_{org}$ of organic matter in the wheat straw biomass-amended soil was clearly less stable than that of biochar, and accordingly it was inferred that some labile C was trapped in carbonate, which also can explain why the signal of $\delta^{13}C_{carb}$ from the soil carbonate was slightly more negative in the lysimeter with wheat straw pellets. The trend in carbon isotope values for the landscaping soils in the incubated environments partially demonstrated that biochar positively impacted carbon retention in the land.

Except soil carbon, other forms of carbon at two different lysimeters showed various changes and relationships with soil amendments. Firstly, the concentration of dissolved total carbon (DTC) as well as the total carbon loss due to leaching, were similar in both environments, whereas the concentration of dissolved organic carbon (DOC) in lysimeter BC was significantly lower by 42% than for lysimeter WP. The alkalinity and pH in lysimeter BC were higher, though a significant statical test outcome was found in pH alone. Secondly, again, the difference of vegetation yield between the biochar-amended soils and biomass pellet-amended soils was small over three consecutive years. Furthermore, both the manually-installed tubes method and EGM-4 Gas Monitor all measured a continuous fluctuation of CO_2-C emissions at the two lysimeters throughout the whole experimental period, which showed the difficulty of monitoring CO_2-C flux at these outdoor facilities without automation in the long-term. Alternatively, we could consider the consecutively daily soil CO_2 gas measurements at the initial fortnight of application, which would be better if gas sampling is conducted at the same time every day, and later adjust the frequency of CO_2 measurements to once a week or a month. However, we could estimate the total carbon emitted from soil respiration based on the difference between the carbon addition of soil amendments and other carbon change pathways such as C gain (soil, grass) and C decrease (leachate). From the estimated results, it became apparent that CO_2 emissions were mainly responsible for substantial losses of carbon, rather than leaching, which was most notable in the lysimeter with wheat straw pellets added. Overall, the three-year *in situ* lysimeter study demonstrated that biochar could be a feasible option to enhance soil carbon sequestration as well as remediating soil environments. The final soil carbon accumulation achieved was 2.91±2.48 $Kg\cdot m^{-2}$ with the introduction of 2% w/w wheat straw biochar. This corresponds to the potential ability of a 1-hectare Newcastle urban green space in terms of carbon sequestration at around 30 tonnes. The increase of carbon stock among landscaping soils due to the introduction of biochar, either in the top 0-15 cm soils or 15-40 cm soils, highlighted the importance of taking biochar into consideration for actively improving the terrestrial carbon pool, and moreover biochar could add “value” in multiple ways such as recycling wastes, soil conditioner, adjusting urban drainage, reducing stormwater runoff and filtering contaminants in stormwater, etc. However, the biochar production at the institutional scale is less likely achievable due to insufficient feedstock, according to the farm director in Newcastle University; at the same time, the considerations of biochar deployment in terms of operation costs, additional labour care and practical safety, from either the perspective of estate manager or sustainability team, importantly determine the potential use of biochar in the university.

6.4 The potential of terrestrial carbon augmentation methods for helping achieve Newcastle University’s ambitious net-zero carbon goals

Making full and efficient use of terrestrial resources to maximize the carbon sequestration potential is a supplementary method for the current on-going carbon management plan of Newcastle University which has implemented other programmes, including energy saving, better building management, improved allocation of academic resources, and appealing for behaviour change, etc. Considering Newcastle University has proposed to meet net-zero carbon by 2030, but will likely still be accountable for some carbon emissions in nine years’ time, this project can offer the following scenarios for carbon offsetting in the institutional estate:

1. Across agricultural land at the two farms, the transformation offsetting the largest amount of carbon for a short period of time (5 years) was to change the current agricultural land use back to how CPF was managed one hundred years ago: expanding the percentage of permanent grassland from 21% to 84% whilst decreasing the arable land from 79% to 16%, and this would sequester 64% of annual carbon emissions of Newcastle University (6,406 tonnes of CO_2 equivalents-C) (Newcastle University, 2021) for a period of five years;
2. When afforestation is considered, the most substantial approach to store more carbon into the soil and tree biomass of the institutional estate is converting all agricultural land at the 2 farms to coniferous woodland, which could accomplish a 50% offsetting of annual institutional carbon emissions over a period of 40 years; alternatively, another perhaps realistic scenario is converting NF into a forestry research centre with the mixed woodland composed by 50% of coniferous and 50% broadleaved, which can sequester 29% of carbon emitted from Newcastle University per annum over 40 years;
3. For Newcastle University’s city campus, planting in all green spaces the 4 tree species with the highest capability of carbon sequestration (Ash, Swedish Whitebeam,

Sycamore, Large-leaved Lime), would result in only a minor amounts of carbon sequestration every year - only 1.14%;

4. Finally, if we consider deploying wheat straw biochar in the topsoil of the green areas of the campus overall (the 25 ha of city lands including the 16 ha of the four sports grounds and the 9 ha of green space surrounding academic buildings) at a rate of 2% w/w organic carbon augmentation, this action could bring about carbon offset of around 727 tonnes, accounting for 11% of the annual CO_2 equivalents -C emissions of Newcastle University.

From these scenarios it quickly becomes evident how Newcastle University, located in the city core and sharing multiple urban functions with many other interested parties, meets a big challenge to off-set tangible amounts of its current carbon emissions by land use transformation in its urban estate alone. However, implementing land conversion in rural areas, especially the development of forests and pastures, can substantially assist in achieving net-zero carbon, but requires substantial change to the land use at the two research farms. Carbon-offsetting is too often perceived as an abatement measure which can be readily implemented overseas, typically in low- or middle-income countries, against payments for the related services. Off-setting institutional emissions in "one's own backyard", i.e. in Newcastle University's rural estate, brings some of the related challenges, trade-offs, and conflicts, much closer to home. This was for example reflected in the cautious response of the farm director to the proposed ideas in an interview, which highlighted some of the real-world challenges, incl. tenancy contracts, commercial considerations, and practical challenges in implementing biochar-based carbon sequestration.

In addition to offsetting carbon emissions overseas, utilising peatland is a way to increase terrestrial carbon stock as it has the highest carbon density among all land types, which has been broadly acknowledged (IUCN, 2018). The area of peatland in the UK is 3×10^6 ha approximately, ranking in the top 10 countries globally (IUCN, 2018), but 80% of peatland in the UK is damaged and so needs immediate conservation to maintain the original ecosystem benefits such as habitants of wildlife, keeping biodiversity, infiltering rainfall, remediating flooding, balancing the carbon emissions caused by other conflicting land managements, etc (IUCN, 2018). Protecting carbon stores by restoring peatland is an economical means to optimize positive environmental outcomes and establish more employment opportunities to local people, particularly when this work could develop pluviculture in lowland peatland and commercial forestry in upland peatland, and further maybe encourage tourism. Urban institutions like Newcastle University, or Newcastle City Council, could establish regional partnerships with rural councils and landowners to off-setting urban carbon emissions by restoring peatlands. Also, enhanced weathering with the Urey reaction between carbonic acid and certain rock materials, is a possible mitigation measure to speed the CO_2 capture as well as slow global warming. A project organized by Beerling et al. (2020), building a performance model to evaluate the potential of carbon dioxide removal by enhanced weathering across the cropland over 12 countries, has acknowledged that the agricultural land can be greatly fertilized by the implementation of enhanced weathering with the multiple mineral nutrients. Furthermore, the enhanced weathering has a high potential for amending deteriorated agricultural soils, which may address the food shortage over six billion people (Beerling et al., 2020).

6.5 Future Work

This study has confirmed that terrestrial land can make a significant contribution with respect to carbon sequestration. However, some questions remained at the end of this study due to limited time and resources, and these need to be considered in more detail in the future. The following lists the aspects waiting to be researched.

The classification of agricultural land use is simplified in this project, as its final expressions of “permanent grassland” and “arable land” in Chapter 3. This is because in our study different crops cultivated in the same field have a differing time frame, which makes it difficult to classify agricultural soil use types explicitly, and further influences the precision of carbon stock estimation over the agricultural land at farms. In the future, to monitor how the crop choice would change soil carbon storage, the carbon practitioners could negotiate with farm directors to alter some agriculture lands regularly or cropping one field over a long term (at least over 5 years). In that case, the impact of different crops on 0-90 cm soil carbon stock would gain a better understanding. Besides, the number of soil cores sampled under the measured biomass areas in woodlands is few, which results in the soils only being classed with coniferous or broadleaved instead of considering different tree species. It is recommended that more soils are collected under the same tree species, and the distance between the soil core and tree trunk could also be measured. Similarly, for obtaining a full understanding of soil carbon distribution in different urban land covers, not only in urban greenspace, soil samples under paving constructions and tarmac should be collected, which would be useful for analysing the soil carbon transportation below cities.

"Climate Action Plan: Phase 1" of Newcastle University sits alongside the existing carbon management plan, which is already sending academic sectors positive signals by demonstrating that it is possible to cut emissions even when undertaking the expansion of campus size and increasing enrolment. However, achieving the considerable amounts of carbon sequestration over the city campus by transferring land use is challenging, based on the practical perspectives from institutional managers. Thus, the university needs more involvement of its farmlands in the rural areas. This study has suggested several land transformations; it is worth starting a small-scale field trial at the farms to build confidence in the practical feasibility of such carbon sequestration. When a new woodland is developed, annual measurements of soil carbon change and tree growth are necessary. This will help land managers to better understand the real terrestrial carbon accumulation due to the land conversion in north-eastern England and how these trends will fluctuate over time. Such fieldwork should also quantify how greenhouse gas emissions from the terrestrial environment beyond carbon dioxide, such as methane or nitrous oxide gas emissions respond to the change in land management.

The lysimeter study conducted in this study should continue over an extended period of one or more decades, and the monitoring could be optimized further: the observation of CO_2-C emissions should be shortened to daily when the soil amendments have been applied, with an increase in the interval of gas sampling times when results start to show only minor fluctuations; the leachate quality investigated during the past three years was limited to the carbon concentration, pH and alkalinity, but the concentrations of metals and anions should also be assessed. It would also be valuable to compare the biodiversity of the lysimeter ecosystems for the different amendment strategies. Additionally, a similar study could consider cropping C4 plants in soils amended with C3 plant-biochar, or C3 plants in soils amended with C4 plant-biochar. The difference of $\delta^{13}C$ between C3 and C4 plants is extremely apparent, so the $\delta^{13}C$ of soils would show a distinct value making it easier to analyse the composition of soil organic carbon if the method mentioned is carried out. Also, the experiment could be redesigned for different purposes. For instance, applying biochar into the soils already managed as lawns, as this may represent a practical soil carbon change occurring more widely over the urban green space, and not like engineering soils used in our study which are a mixture of anthropogenic and geological materials that are often biologically/chemically-unfriendly for plant growth; or another biochar should be considered as amendment, because the wheat straw residues are completely utilized in the university farms and not enough materials are left to produce the wheat straw biochar with institutional biomass residues. If land use transformations at the university farms can be implemented, adding biochar into those soils may offer an opportunity to compare the biochar stability between the human-built equipment and a more natural environment.

The methodological framework developed in this study to quantify the terrestrial carbon and its sequestration potential for various scenarios is a worthwhile reference for other institutions with large landholdings, such as local authorities, nature reserves, the supply base of retail industries and water companies, etc. However, our results only involve a limited number of current land types, and the composition of land types managed by others maybe more complicated. Extending the work to other institutions would allow carbon practitioners to investigate the actual soil carbon storages and off-setting opportunities in settings not yet discussed in this project.

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Appendix A

Appendix A1. Carbon management plans of academic institutions

Carbon management plans of 16 universities: Out of sixteen universities reviewed, only two (The State University of New York at Buffalo and the University of Tasmania) have considered land management as a part of their carbon management plan (Table A. 1). No university has established a baseline of the current terrestrial carbon stock in their land as part of their carbon management plan. The University of Tasmania emphasized the importance of maintaining biodiversity and reducing the threat from exotic plants to native plant species on their land; The State University of New York at Buffalo proposed to explore the possibility of growing biofuels on their land.

Higher Education Institution	CO_2 Emission Target ‡	Awareness Campaigns; Behavioural Change (virtual meeting…); Teaching	Transportation (Cycling, university-owned fleet, reduce business travel...)	Energy saving or replacement (Heat, water, voltage, electricity, lighting), deployment of renewables (wind, solar, tide...)	Green IT (Multifunctional printing, upgrading the server…)	Building Refurbishment (Disposal of old buildings, double glazing...)	Waste recycling and management (Office paper…)
Cardiff University	Carbon neutral (scope1&2 §) by 2023	√	√	√		√	√
Durham University	19085 tonnes by 2020	√	√	√	√	√	√
Loughborough University	16818 tonnes by 2020	√		√	√	√	
Newcastle University	23694 tonnes by 2020 (CO_2 equivalents-C)	√	√	√	√	√	√
The University of Nottingham	41000 tonnes by 2020	√		√	√	√	
The University of Sheffield	19306 tonnes by 2020/21	√	√	√		√	
University of Bath	29167 tonnes by 2020	√	√	√	√	√	√
The University of Edinburgh	Carbon neutral by 2040	√	√	√	√	√	√

Higher Education Institution	CO_2 Emission Target ‡	Awareness Campaigns; Behavioural Change (virtual meeting…); Teaching	Transportation (Cycling, university-owned fleet, reduce business travel...)	Energy saving or replacement (Heat, water, voltage, electricity, lighting), deployment of renewables (wind, solar, tide...)	Green IT (Multifunctional printing, upgrading the server…)	Building Refurbishment (Disposal of old buildings, double glazing...)	Waste recycling and management (Office paper…)
University of Glasgow	55000 tonnes by 2020/21	√	√	√	√	√	√
University of Leicester	Scope 1 &2 § was reduced by 25% by 2020	√	√	√	√	√	√
University of Liverpool	37391 tonnes by 2020	√	√	√	√	√	√
University of York	Absolute reduction target of 10764 tonnes for 2020	√	√	√	√	√	
University of Lincoln (Nebraska)	*Scope 1&2 § was reduced by 43% by 2020/21*	√	√	√	√	√	√
The University of British Columbia	*Net-positive performance in operational carbon by 2050*	√	√	√	√	√	√
The University of Tasmania	*Carbon neutral certified since 2016*	√	√	√	√	√	√
The State University of New York at Buffalo	*Climate Neutral by 2030*	√	√	√	√	√	√

Higher Education Institution	Monitoring and Surveying	Allocating carbon reduction aim to departments	Sustainable Purchasing Policy	Quantify the carbon impact of food systems	Space Utilisation (Efficient timetabling, use of zoning in buildings…)	Cooperation (local community, city council, national organisations…)	Carbon offsetting (last resort)	Land use Management Programme
Cardiff University	√		√					
Durham University	√	√	√		√	√		
Loughborough University		√			√	√	√	
Newcastle University	√		√					
The University of Nottingham	√							
The University of Sheffield	√	√	√			√		
University of Bath	√		√					
The University of Edinburgh	√		√	√		√		
University of Glasgow					√			
University of Leicester	√	√	√		√	√	√	
University of Liverpool	√		√	√	√			
University of York	√		√					
University of Lincoln (Nebraska)	√	√	√			√		

Higher Education Institution	Monitoring and Surveying	Allocating carbon reduction aim to departments	Sustainable Purchasing Policy	Quantify the carbon impact of food systems	Space Utilisation (Efficient timetabling, use of zoning in buildings…)	Cooperation (local community, city council, national organisations…)	Carbon offsetting (last resort)	Land use Management Programme
The University of British Columbia			√					
The University of Tasmania	√	√	√			√	√	√
The State University of New York at Buffalo	√	√	√	√		√		√

Table A. 1. Summary† of CO_2 emissions reductions strategies of universities in the UK and *overseas* (*Italics*).

†: The titles and projects inventories of carbon management plans from every university are different, and some strategies overlap with others, and the descriptions of each item in the top row are concluded by the author.
‡: The standards of CO_2 emissions reduction targets are different in various universities and countries. The details of CO_2 emission baseline and reduction plans can be found from their own carbon management plans in the references.
§: Given the difficulties on calculating and managing the indirect CO_2 emissions from mobile sources, some universities only set the CO_2 emission reduction target according to the stationary sources. For the explanations of scope 1& 2 emissions, see GREENHOUSE GAS PROTOCOL (2019).

Appendix A2. Supplemental method details

Soil sample processing and analysis: Soil sample processing and analysis: The samples were analysed for carbon as percent mass by a dry oxidative combustion procedure at up to 1000 ℃ using the LECO RC 612 analyser (LECO Corporation (2018),Saint Joseph, Michigan USA). This combustion analysis assumed that organic carbon is evolved by decomposition of organic matter to CO_2 between 150℃ to 450℃, and inorganic carbon at temperatures between 450℃ to 1000℃, broadly in line with the detailed thermal characterization carried out by Lopez-Capel et al. (2005). Thermal analysis is more reliable than acidification when distinguishing organic from inorganic carbon (Siavalas et al., 2013). Ex-situ bulk density was calculated in the laboratory by considering the dry soil mass obtained on average for the core volume from each soil depth layer, and the soil carbon density ($Kg \cdot m^{-3}$) was calculated from the carbon content per gram dry soil multiplied by the ex-situ bulk dry soil density ($Kg \cdot m^{-3}$):

$$\textit{Total C density (Kg C}\cdot m^{-3}\textit{) = C concentration (Kg C/ Kg soil)} \times \textit{soil density (Kg}\cdot m^{-3}\textit{)}$$

Equation A1

The carbon storage per land surface area ($Kg\ C \cdot m^{-2}$) in the soil sampled by coring down to 90 cm depth was the sum of carbon storage per unit area in each 30 cm soil layer:

$$\textit{Total C storage over 90 cm soil depth (Kg C}\cdot m^{-2}\textit{) = [C density at 0-30 cm layer (Kg C}\cdot m^{-3}\textit{)} \times 0.3m\textit{] + [C density at 30-60 cm layer (Kg C}\cdot m^{-3}\textit{)} \times 0.3\ m\textit{] + [C density at 60-90 cm layer (Kg C}\cdot m^{-3}\textit{)} \times 0.3m\textit{]}$$

Equation A2

In addition, soil pH was measured in a soil-water suspension of 1:2.5 (w/v) (Zani et al., 2020, Reynolds et al., 2013). Grounded soil samples (5 g) were mixed with deionised water (12.5 mL), then stirred with a magnetic stirrer for 30 minutes, and then leaving the solution to settle for 1 hour before measuring soil pH with a pH Meter (900 multiparameter water quality meter, BANTE Instrument, Shanghai, China).

Soil carbon mapping: Carbon distribution maps were made using ArcMap (version 10.6.1) with geostatistical analysis extension. Ordinary Kriging was used as a linear geostatistical interpolation technique, which can provide estimates for unsampled points according to weighted sums of the adjacent sampled concentrations (Panday et al., 2018, Shit et al., 2016). In this research, experimental semi-variograms using the Gaussian model with two to six neighbourhood information were chosen, because they showed the best fit among all candidate models evidenced by the lowest root mean square error and mean estimation error closest to zero (Chabala et al., 2017).

Tree and tree canopy coverage survey: The parameters tree diameter at breast height (DBH, measured 1.3 m from the ground), height, and species were obtained for all trees within 6 surveying plots in woodlands with a total of 60 coniferous trees and 57 broadleaved trees at Cockle Park Farm, and 30 coniferous trees at Nafferton Farm in July and October 2019, respectively. Tree DBH and height were measured using tape and VERTEX IV (HAGLÖF, Bromma, Sweden). The software package i-Tree (2020) from the United States Department of Agriculture Forest Service enabled quantification of carbon in woodland biomass. The tool i-Tree Eco required DBH, height, tree species, the geographic location and a weather station as the main input parameters to determine carbon storage and carbon sequestration for different trees (Raum et al., 2019, Rocco et al., 2018). Five relevant species were selected for Cockle Park Farm to predict the carbon storage in trees: European Larch (*Larix decidua*), Sycamore (*Platanu*), English Oak (*Quercus Robur*), Sitka Spruce (*Picea Sitchensis*), and Norway Spruce (*Picea abies*). Sitka Spruce (*Picea Sitchensis*) and Norway Spruce (*Picea abies*) were evaluated in the Nafferton Farm woodland. For comparison with i-Tree Eco, the Woodland Carbon Code: Carbon Assessment Protocol of the Forestry Commission of England (Jenkins et al., 2018) was also used to estimate the carbon storage of trees. The tool i-Tree Canopy complements i-Tree Eco by estimating land tree cover (Brent, 2014, Rogers and Jaluzot, 2015, UBOC, 2020). In this study, i-Tree Canopy provided the percentage of tree canopy coverage in the woodlands, considering that there were small areas covered by grass and shrubs within each woodland at both farms. 200 random sampling points were chosen from aerial images of the woodlands at each farm for estimating the tree cover area. The carbon stored by trees in the woodlands was calculated by multiplying the whole area of tree cover obtained from i-Tree Canopy, and the carbon storage of the trial plots obtained from i-Tree Eco:

Tree carbon stocks in woodlands (Kg C) =The whole area of trees (m^2) ×The average of carbon density in the surveyed plots (Kg C·m^{-2}) *Equation A3*

Because the exact time of tree planting was unknown, the average stem radial increment index (De Vries, 1987, Smith and Shifley, 1984) was used instead to estimate the tree age from DBH. The average tree age was estimated to be 40 years, which was then used to estimate the carbon sequestration capacity of trees in newly planted woodland over a period of 40 years on an annual basis (Kg·m^{-2}-yr). Also, individual trees and small groups of trees grew along field edges and in some fields at the two farms. The crown area of these trees was estimated on satellite images on Google Earth, and multiplied with the mean carbon storage (Kg C/m^2 crown area) of the woodland trees at Cockle Park and Nafferton Farm, respectively, to calculate the carbon stock of these trees.

Statistical data analysis of land management effects: Institutional crop rotation records at the two farms were available from Gatekeeper (2020), a software package for farm management. Crops included: grass, winter wheat, winter barley, oil seed rape, spring barley, white clover ley, red clover ley. Fields at Cockle Park Farm could be divided into two land management groups based on crop records from 2016 to 2020: permanent grassland and arable land. At Nafferton Farm, management between 2002 and 2017 divided the farm into a conventional and organic system, so the fields at the farm were classified in line with these two farm treatments. Although classification according to crop records was made at Nafferton Farm, only two fields were managed as permanent grassland with two soil cores collected in our study. Woodland at Cockle Park Farm could be distinguished according to the time of establishment. In the north-eastern area, Digimap (2020) showed a woodland established after the 1960s, while the woodland on the western part of the farm has been in place for at least 100 years. 10 sampling locations in the older woodland, and 6 sampling locations in the more recently established woodland were available at Cockle Park Farm. At Nafferton Farm, woodland existed at the south-eastern corner of the farm since at least the 1860s according to historic maps (Digimap, 2020, MAGIC, 2020). The woodland maps on MAGIC classified them as coniferous or broadleaved by polygon areas: 6 and 11 soil sampling locations were in broadleaved areas, versus 10 and 5 in coniferous areas, at Cockle Park and Nafferton, respectively.

The responses between means of continuous variables (total carbon, organic carbon, pH) to variations of independent factors (as an individual group) such as land management or depth were tested by One-way ANOVA. While the relationship between different classes in the same group (e.g., 30 cm depth -60 cm depth, 60 cm depth-90 cm depth) and the interacted influence caused by multiple groups were processed by the Tukey's HSD test in univariate analysis of SPSS (IBM SPSS Statistics 26, Armonk, New York, USA). All these differences were considered significant for a p-value < 0.05. Using the average amount of soil and tree carbon per m^2 of surface area for each land use type to predict the total carbon stocks at the two farms as a function of land management (Table A. 8-11).

Within Single Farm		CPF			NF		
		TC	TOC	pH	TC	TOC	pH
Agricultural Land	Depth	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	Fields Classification†	0.041	0.008	0.180	0.717	0.718	<0.001
	Depth x Fields Classification	0.004	0.003	0.365	<0.001	<0.001	0.327
Woodlands	Depth	<0.001	<0.001	<0.001	<0.001	<0.001	0.003
	Woodland Age	0.397	0.401	0.084	n.a §	n.a §	n.a §
	Leaves Types	0.283	0.279	0.369	0.886	0.808	0.935
	Depth x Leaves Types	0.455	0.461	0.679	0.511	0.531	0.577
Agricultural Land & Woodlands	Depth	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	Land Use‡	<0.001	<0.001	0.069	<0.001	<0.001	<0.001
	Depth x Land Use	0.023	0.031	<0.001	0.724	0.862	0.100

Table A. 2 Significant difference (One-way ANOVA, p-value) results for statistical analysis of soil carbon results and pH in agricultural land and woodlands caused by different independent factors at Cockle Park (CPF) and Nafferton Farm (NF), respectively.

TC: total carbon: TOC: total organic carbon.

Significance was acknowledged when p<0.05.

†: Field classification refers the soil classification according to the vegetation rotation records at CPF and the division of organic/conventional area at NF.

‡: Land use in this part refers to the soil samples from the agricultural land or woodlands at each farm.

§: no woodland age division at Nafferton Farm.

			Agricultural Land		Woodland		Agricultural Land & Woodland	
			TC	TOC	TC	TOC	TC	TOC
CPF	0-30 cm	30-60cm	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
		60-90cm	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	30-60 cm	60-90cm	0.653	0.462	0.651	0.654	0.41	0.388
NF	0-30 cm	30-60cm	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
		60-90cm	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	30-60cm	60-90cm	0.675	0.192	0.551	0.599	0.454	0.166

Table A. 3 Results of multiple comparisons (Tukey's HSD in univariate analysis: *p*-value) of soil carbon between each paired depth layer in Cockle Park (CPF) and Nafferton Farm (NF), respectively. TC: total carbon: TOC: total organic carbon

	Agricultural Land		Woodlands		Agricultural Land & Woodlands	
	TC	TOC	TC	TOC	TC	TOC
Farm location†	0.003	0.089	0.155	0.259	0.001	0.030
Soil type classification† §			0.799	0.708	<0.001	<0.001
Soil type classifications‡ § x Depth			<0.001	<0.001	<0.001	<0.001
Soil type classifications‡ § x Farm location			0.219	0.331	0.007	0.119
Depth x Farm location ‡	0.035	0.017	0.976	0.970	0.264	0.288
Depth x Farm x Soil type classification‡ §			<0.001	<0.001	<0.001	<0.001

Table A. 4 Significant difference (*p*-value) results for statistical analysis of soil carbon results in agricultural land and woodlands caused by different independent factors between Cockle Park (CPF) and Nafferton Farm (NF).
TC: total carbon: TOC: total organic carbon.
†: The significant difference is obtained by One-Way ANOVA for top two rows.
‡: The significant difference is obtained by Tukey's HSD in univariate analysis for the rest of four rows.
§: No soil type classification in agricultural land where cells are blank, because the principles of categorizing fields at two farms are different. Soil type classifications in woodlands where cells are filled with blue means the division of soil samples between broadleaved woodland and coniferous woodland. Soil type classifications in the grey cells means the division of soil samples between agricultural land and woodlands.

	0-30cm		30-60cm		60-90cm	
	TC	TOC	TC	TOC	TC	TOC
CPF						
Tests between permanent grassland and arable land	0.004	0.002	0.918	0.962	0.97	0.925
NF						
Tests between organic and conventional area	0.275	0.348	0.317	0.282	0.658	0.743

Table A. 5 Results of multiple comparisons (One way ANOVA: p-value) of soil carbon between permanent grassland/ arable land in Cockle Park (CPF) and between organic/conventional area at Nafferton Farm (NF), respectively.
TC: total carbon: TOC: total organic carbon.

				Agricultural Land					
				TC			TOC		
				0-30cm	30-60cm	60-90cm	0-30cm	30-60cm	60-90cm
CPF	Woodlands	TC	0-30cm	*					
			30-60cm		**				
			60-90cm			*			
		TOC	0-30cm				*		
			30-60cm					**	
			60-90cm						**
NF		TC	0-30cm	**					
			30-60cm		***				
			60-90cm			***			
		TOC	0-30cm				***		
			30-60cm					***	
			60-90cm						***

Table A. 6 The significant difference (Tukey's HSD in univariate analysis) of soil carbon results in agricultural land and woodlands for the same collection layers in Cockle Park Farm (CPF) and Nafferton Farm (NF), respectively.

* Significant at $p<0.05$; ** Significant at $p<0.01$; ***Significant at $p<0.001$

	0-30cm		30-60cm		60-90cm	
	TC	TOC	TC	TOC	TC	TOC
CPF agricultural land	-0.119	-0.133	0.009	-0.092	0.033	-0.216
CPF Woodland	-0.475	-0.476	0.014	0.002	-0.525*	-0.41
CPF Total	-0.504**	-0.511**	-0.365**	-0.410**	-0.252	-0.441**
NF agricultural land	-0.362*	-0.334	-0.093	-0.140	0.250	0.135
NF Woodland	-0.357	-0.369	-0.022	-0.166	-0.509*	-0.560*
NF Total	-0.440**	-0.472**	-0.338*	-0.415**	-0.227	-0.533**

Table A. 7. Pearson correlation coefficients between soil pH and soil total carbon (TC), soil organic carbon (TOC), respectively. CPF: Cockle Park Farm; NF: Nafferton Farm.

* Correlation is significant at the 0.05 level
** Correlation is significant at the 0.01 level

		Unit	CPF	NF	Rural estate
Carbon Storage	Soil carbon storage top 90 cm of permanent grassland	$Kg{\cdot}m^{-2}$	12.14	17.13	n.a.
	Soil carbon storage top 90 cm of arable land	$Kg{\cdot}m^{-2}$	10.30	12.16	n.a.
	Soil carbon storage top 90 cm of coniferous woodlands	$Kg{\cdot}m^{-2}$	15.30	16.64	16.13
	Soil carbon storage top 90 cm of broadleaved woodlands	$Kg{\cdot}m^{-2}$	13.25	16.34	n.a
	Biomass carbon storage coniferous woodlands	$Kg{\cdot}m^{-2}$	12.68	12.60	12.63
	Biomass carbon storage broadleaved woodlands	$Kg{\cdot}m^{-2}$	10.65	n.a	10.65
	Biomass carbon per hedgerow tree	Kg	395.00	395.00	395.00
Field Area	Land area permanent grassland	hectares	0.0	0.0	0.0
	Land area arable land	hectares	0.0	0.0	0.0
	Land area coniferous woodlands	hectares	307.0	498.0	805.0
	Land area broadleaved woodlands	hectares	0.0	0.0	0.0
	Number of hedgerow trees	trees	1146	1260	2406
Carbon stock	Soil carbon in top 90 cm of permanent grassland	tonnes	0	0	0
	Soil carbon in top 90 cm of arable land	tonnes	0	0	0
	Soil carbon in top 90 cm of coniferous woodlands	tonnes	46962	82872	129834
	Soil carbon in top 90 cm of broadleaved woodlands	tonnes	0	0	0
	Biomass carbon in coniferous woodlands	tonnes	38938	62748	101686
	Biomass carbon in broadleaved woodlands	tonnes	0	0	0
	Biomass carbon in hedgerow trees	tonnes	452.7	497.7	950
	Total terrestrial carbon	**tonnes**	**86352**	**146118**	**232470**
Timeframe †		years	40		
Total extra carbon sequestration ‡		tonnes	128850		
Annual carbon sequestration §		tonnes/year	3221		
Ratio of annual carbon sequestration to the current institutional C emissions			0.50		

Table A. 8 Scenario 1 for carbon sequestration: convert all land at two farms to coniferous woodland. CPF: Cockle Park Farm: NF: Nafferton Farm.

†: The age of mature tree is estimated as 40 years based on the relationship formula between tree DBH and tree age.

‡: Total extra carbon sequestration is the difference between the estimated carbon storage (232470 tonnes) and the current institutional carbon storage (103620 tonnes, Table 3.1).

§: Annual carbon sequestration is obtained by equation: $\frac{Total\ extra\ carbon\ sequestration(tonnes)}{40\ years}$.

		Unit	CPF	NF	Rural estate
Carbon Storage	Soil carbon storage top 90 cm of permanent grassland	$Kg \cdot m^{-2}$	12.14	17.13	12.14
	Soil carbon storage top 90 cm of arable land	$Kg \cdot m^{-2}$	10.30	12.16	10.30
	Soil carbon storage top 90 cm of coniferous woodlands	$Kg \cdot m^{-2}$	15.30	16.64	16.55
	Soil carbon storage top 90 cm of broadleaved woodlands	$Kg \cdot m^{-2}$	13.25	16.34	16.26
	Biomass carbon storage coniferous woodlands	$Kg \cdot m^{-2}$	12.68	12.60	12.61
	Biomass carbon storage broadleaved woodlands	$Kg \cdot m^{-2}$	10.65	n.a	10.65
	Biomass carbon per hedgerow tree	Kg	395.00	395.00	395.00
Field Area	Land area permanent grassland	hectares	60.2	0.0	60.2
	Land area arable land	hectares	221.8	0.0	221.8
	Land area coniferous woodlands	hectares	18.7	249.0	267.7
	Land area broadleaved woodlands	hectares	6.3	249.0	255.3
	Number of hedgerow trees	trees	1146	1260	2406
Carbon stock	Soil carbon in top 90 cm of permanent grassland	tonnes	7306	0	7306
	Soil carbon in top 90 cm of arable land	tonnes	22854	0	22854
	Soil carbon in top 90 cm of coniferous woodlands	tonnes	2865	41436	44301
	Soil carbon in top 90 cm of broadleaved woodlands	tonnes	831	40674	41506
	Biomass carbon in coniferous woodlands	tonnes	2375	31374	33749
	Biomass carbon in broadleaved woodlands	tonnes	668	26519	27187
	Biomass carbon in hedgerow trees	tonnes	452.7	497.7	950
	Total terrestrial carbon	**tonnes**	**37352**	**140500**	**177852**
Timeframe†		years	40		
Total extra carbon sequestration ‡		tonnes	74232		
Annual carbon sequestration §		tonnes/year	1856		
Ratio of annual carbon sequestration to the current institutional C emissions			0.29		

Table A. 9 Scenario 2 for carbon sequestration: convert Nafferton Farm to 50% broadleaved and 50% coniferous woodland. CPF: Cockle Park Farm: NF: Nafferton Farm.

†: The age of mature tree is estimated as 40 years based on the relationship formula between tree DBH and tree age.

‡: Total extra carbon sequestration is the difference between the estimated carbon storage (177,852 tonnes) and the current institutional carbon storage (103,620 tonnes, Table 3.1).

§: Annual carbon sequestration is obtained by equation: $\frac{Total\ extra\ carbon\ sequestration(tonnes)}{40\ years}$

		Unit	CPF	NF	Rural estate
Carbon Storage	Soil carbon storage top 90 cm of permanent grassland	$Kg\cdot m^{-2}$	12.14	17.13	15.29
	Soil carbon storage top 90 cm of arable land	$Kg\cdot m^{-2}$	10.30	12.16	11.48
	Soil carbon storage top 90 cm of coniferous woodlands	$Kg\cdot m^{-2}$	15.30	16.64	15.76
	Soil carbon storage top 90 cm of broadleaved woodlands	$Kg\cdot m^{-2}$	13.25	16.34	14.29
	Biomass carbon storage coniferous woodlands	$Kg\cdot m^{-2}$	12.68	12.60	12.65
	Biomass carbon storage broadleaved woodlands	$Kg\cdot m^{-2}$	10.65	n.a	10.65
	Biomass carbon per hedgerow tree	Kg	395.00	395.00	395.00
Field Area	Land area permanent grassland	hectares	236.9	407.4	644.3
	Land area arable land	hectares	45.1	77.6	122.7
	Land area coniferous woodlands	hectares	18.7	9.8	28.5
	Land area broadleaved woodlands	hectares	6.3	3.2	9.5
	Number of hedgerow trees	trees	1146	1260	2406
Carbon stock	Soil carbon in top 90 cm of permanent grassland	tonnes	28762	69777	98539
	Soil carbon in top 90 cm of arable land	tonnes	4648	9434	14083
	Soil carbon in top 90 cm of coniferous woodlands	tonnes	2865	1632	4496
	Soil carbon in top 90 cm of broadleaved woodlands	tonnes	831	522	1353
	Biomass carbon in coniferous woodlands	tonnes	2375	1235	3611
	Biomass carbon in broadleaved woodlands	tonnes	668	340	1008
	Biomass carbon in hedgerow trees	tonnes	452.7	497.7	950
	Total terrestrial carbon	**tonnes**	**40602**	**83438**	**124040**
Timeframe ‡		years	5		
Total extra carbon sequestration ‡		tonnes	20421		
Annual carbon sequestration §		tonnes/year	4084		
Ratio of annual carbon sequestration to the current institutional C emissions					0.64

Table A. 10 Scenario 3 for carbon sequestration: convert crop land use at the farm fields back to the split between permanent grassland and arable land shown on the CPF map (from ~ 1900)†: Permanent grassland versus arable land = 84% versus 16%. CPF: Cockle Park Farm: NF: Nafferton Farm.

†: Map of land use at Cockle Park Farm around 1900 can be found in Shiel (2000).

‡: Timeframe is set as 5 years here according to DEFRA (2021). The total extra carbon sequestration is the difference between the estimated carbon storage (124,040 tonnes) and the current institutional carbon storage (103,620 tonnes, Table 3.1). §: Annual carbon sequestration is obtained by equation

$$\frac{Total\ extra\ carbon\ sequestration(tonnes)}{5\ years}.$$

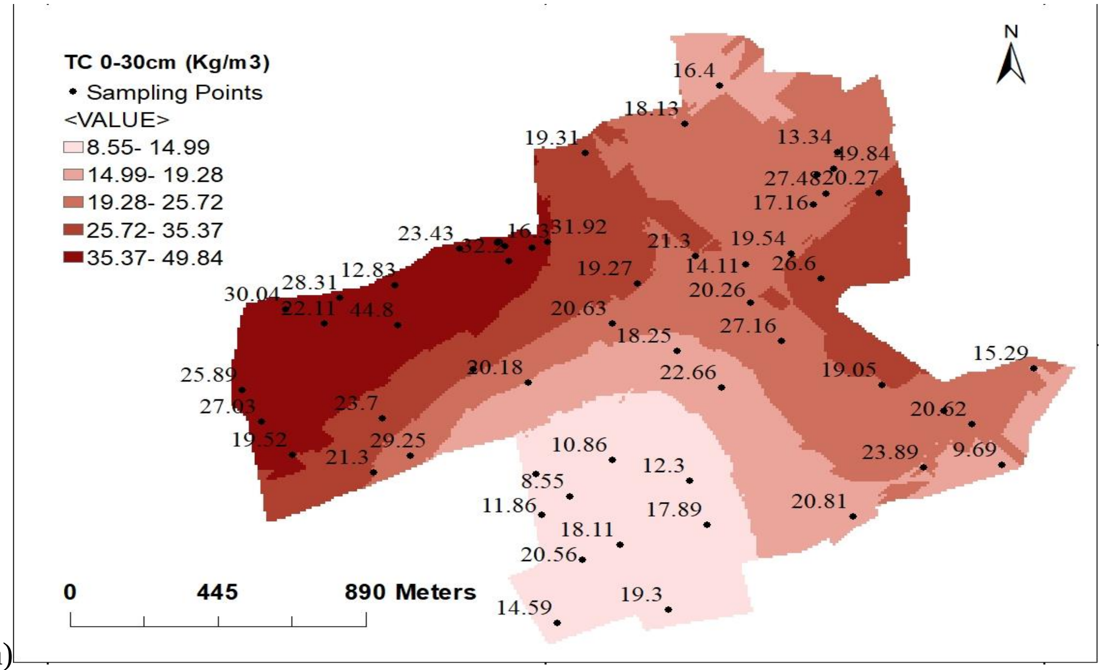
		Unit	CPF	NF	Rural estate
Carbon Storage	Soil carbon storage top 90 cm of permanent grassland	$Kg{\cdot}m^{-2}$	12.14	17.13	14.67
	Soil carbon storage top 90 cm of arable land	$Kg{\cdot}m^{-2}$	10.30	12.16	11.62
	Soil carbon storage top 90 cm of coniferous woodlands	$Kg{\cdot}m^{-2}$	15.30	16.64	15.91
	Soil carbon storage top 90 cm of broadleaved woodlands	$Kg{\cdot}m^{-2}$	13.25	16.34	14.74
	Biomass carbon storage coniferous woodlands	$Kg{\cdot}m^{-2}$	12.68	12.60	12.65
	Biomass carbon storage broadleaved woodlands	$Kg{\cdot}m^{-2}$	10.65	n.a	10.65
	Biomass carbon per hedgerow tree	Kg	395.00	395.00	395.00
Field Area	Land area permanent grassland	hectares	60.2	61.9	122.1
	Land area arable land	hectares	140.3	341.6	481.9
	Land area coniferous woodlands	hectares	59.5	50.6	110.0
	Land area broadleaved woodlands	hectares	47.0	43.9	91.0
	Number of hedgerow trees	trees	1146	1260	2406
Carbon stock	Soil carbon in top 90 cm of permanent grassland	tonnes	7306	10601	17907
	Soil carbon in top 90 cm of arable land	tonnes	14458	41531	55989
	Soil carbon in top 90 cm of coniferous woodlands	tonnes	9098	8413	17511
	Soil carbon in top 90 cm of broadleaved woodlands	tonnes	6231	7178	13410
	Biomass carbon in coniferous woodlands	tonnes	7544	6370	13913
	Biomass carbon in broadleaved woodlands	tonnes	5008	4680	9688
	Biomass carbon in hedgerow trees	tonnes	452.7	497.7	950
	Total terrestrial carbon	**tonnes**	**50097**	**79272**	**129368**
Timeframe ‡		years	40		
Total extra carbon sequestration §		tonnes	25749		
Annual carbon sequestration		tonnes/year	644		

Table A. 11 Scenario 4 for carbon sequestration: offsetting 10% of current annual CO_2-C emissions (641 tonnes) over 40 years, by converting a part of arable land into mixed woodland (50% broadleaved versus 50% coniferous). CPF: Cockle Park Farm: NF: Nafferton Farm.

†: Current CO_2 equivalents-C emissions of Newcastle University in 2019/20 is 6,406 tonnes (Newcastle University, 2021).

‡: The age of mature tree is estimated as 40 years based on the relationship formula between tree DBH and tree age.

§: The total extra carbon sequestration is the difference between the estimated carbon storage (129,368 tonnes) and the current institutional carbon storage (103,620 tonnes, Table 3.1).



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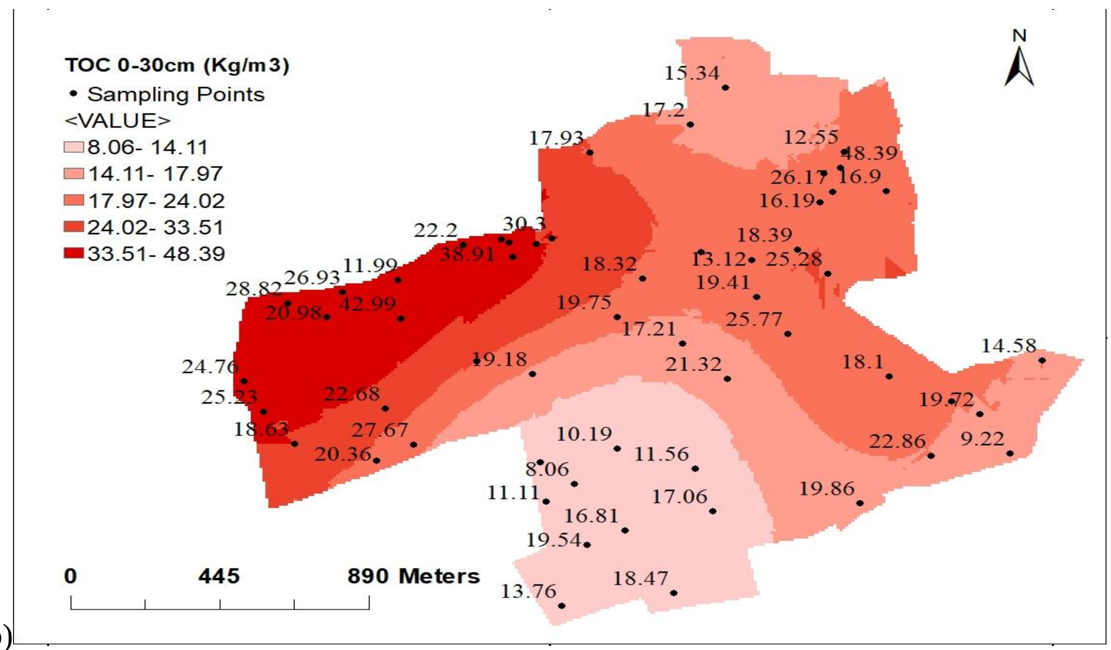


b)

Figure A. 1 Measured carbon and the carbon value estimated by Kriging for 0-30 cm soil at Cockle Park Farm.

The labels aside the sampling plots are the measured carbon value; the classification of counters in the legend shows the estimated value ranges for soil carbon by Kriging. a) total carbon; b) organic carbon.

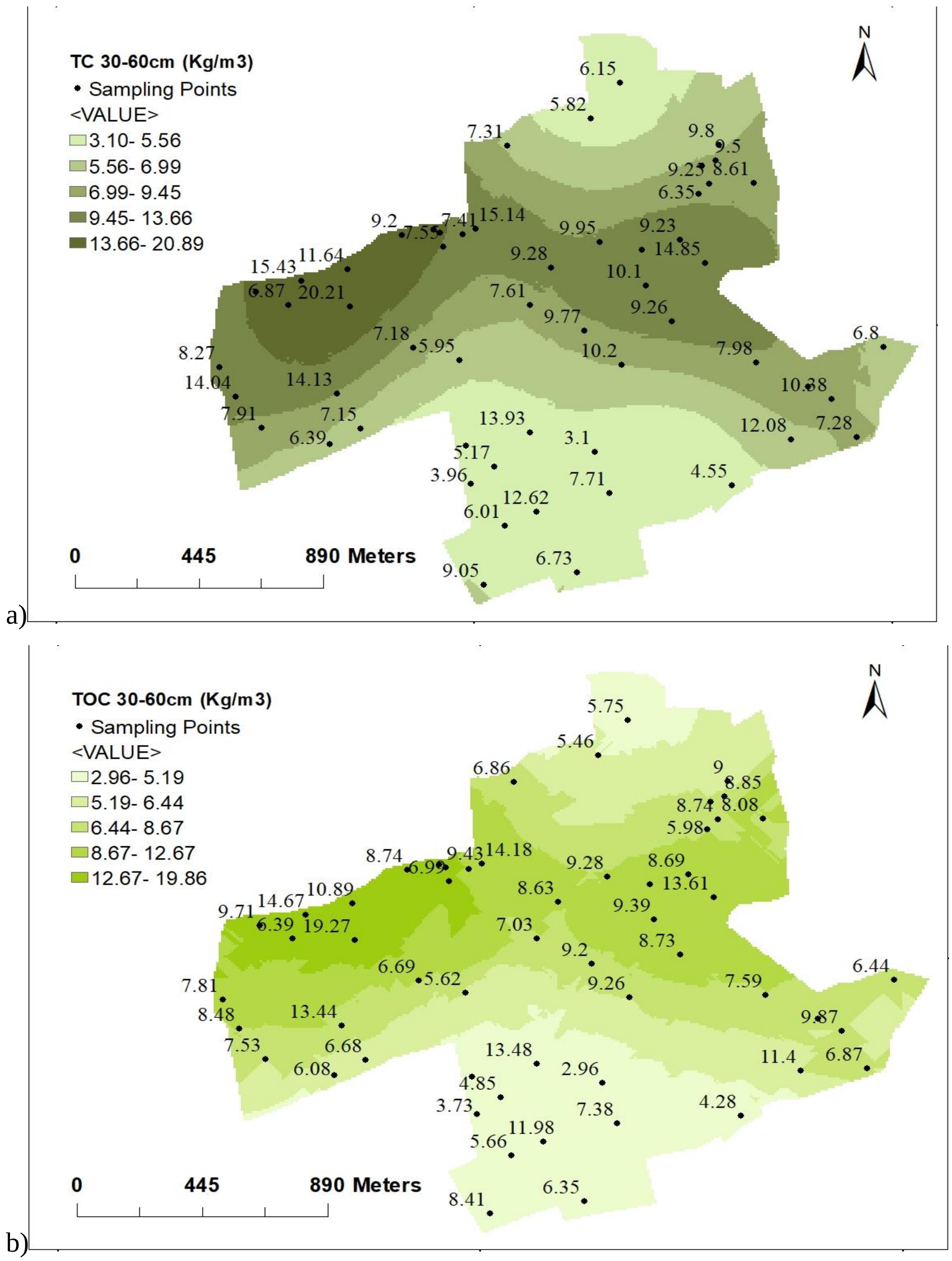
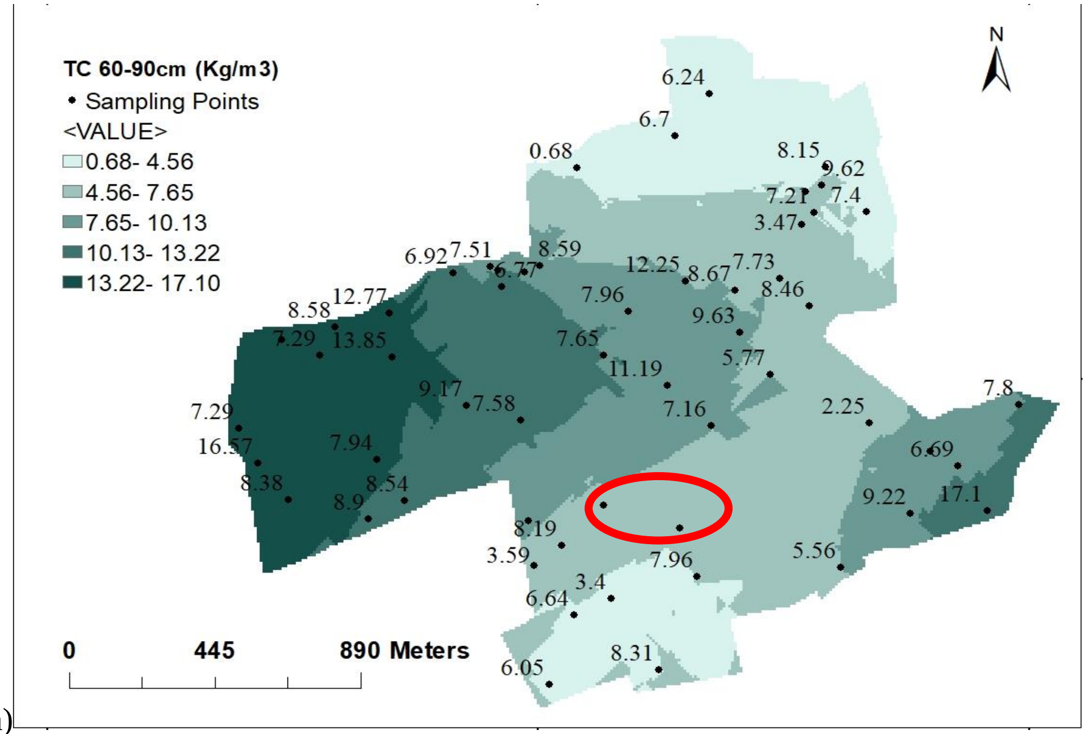


Figure A. 2 Measured carbon and the carbon value estimated by Kriging for 30-60 cm soil at Cockle Park Farm.

The labels aside the sampling plots are the measured carbon value; the classification of counters in the legend shows the estimated value ranges for soil carbon by Kriging. a) total carbon; b) organic carbon.



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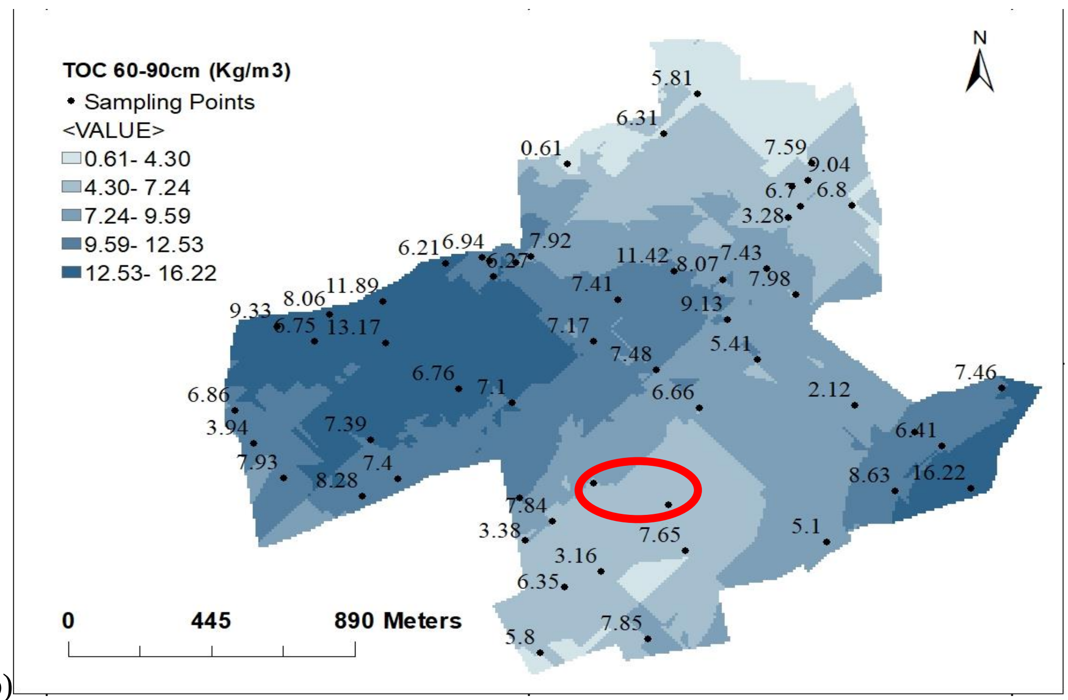
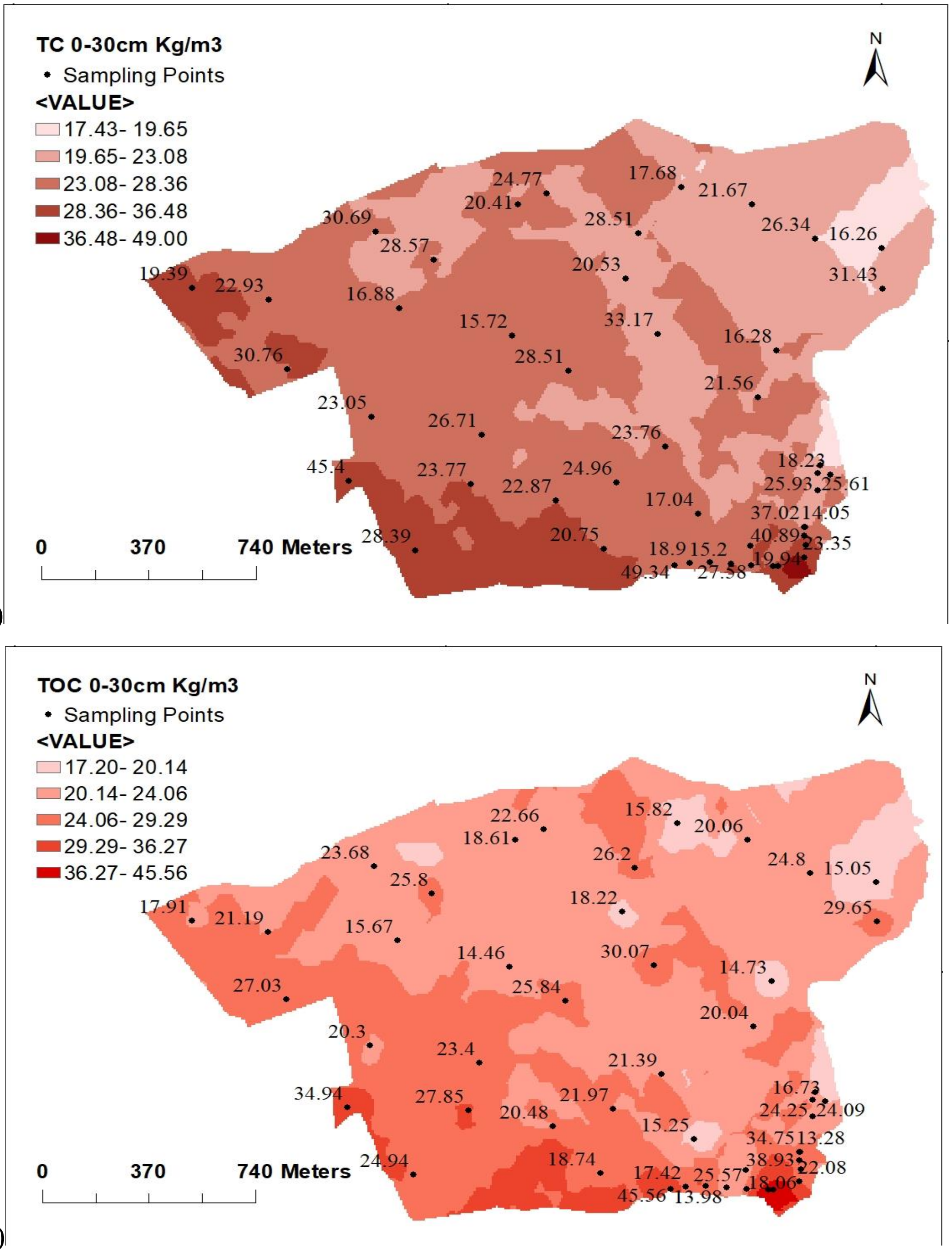


b)

Figure A. 3 Measured carbon and the carbon value estimated by Kriging for 60-90 cm soil at Cockle Park Farm.

The labels aside the sampling plots are the measured carbon value; the classification of counters in the legend shows the estimated value ranges for soil carbon by Kriging. a) total carbon; b) organic carbon. 2 points in the red circle referred the ground conditions which was not suitable for collecting soil samples at 60-90 cm.

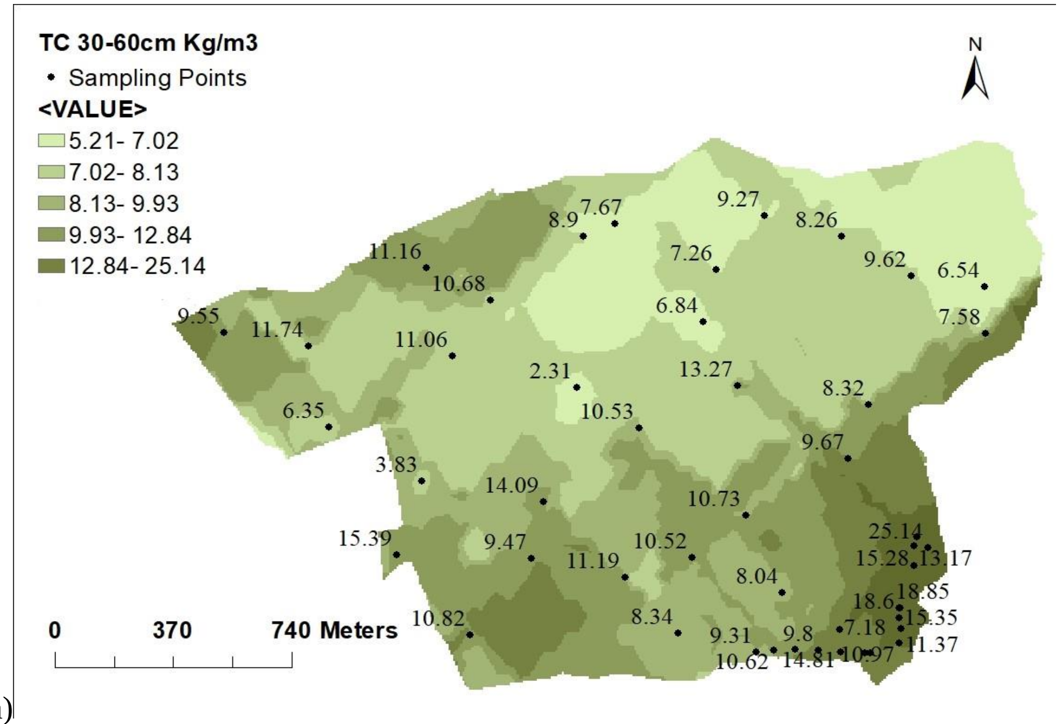


a)

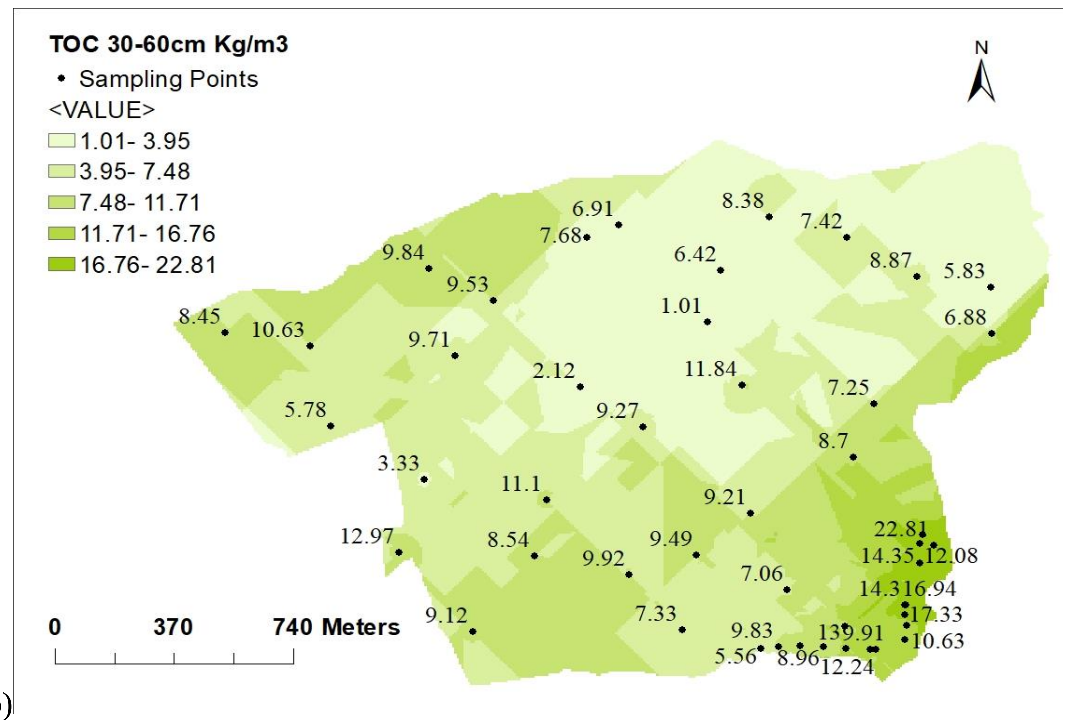
b)

Figure A. 4 Measured carbon and the carbon value estimated by Kriging for 0-30 cm soil at Nafferton Farm.

The labels aside the sampling plots are the measured carbon value; the classification of counters in the legend shows the estimated value ranges for soil carbon by Kriging. a) total carbon; b) organic carbon.



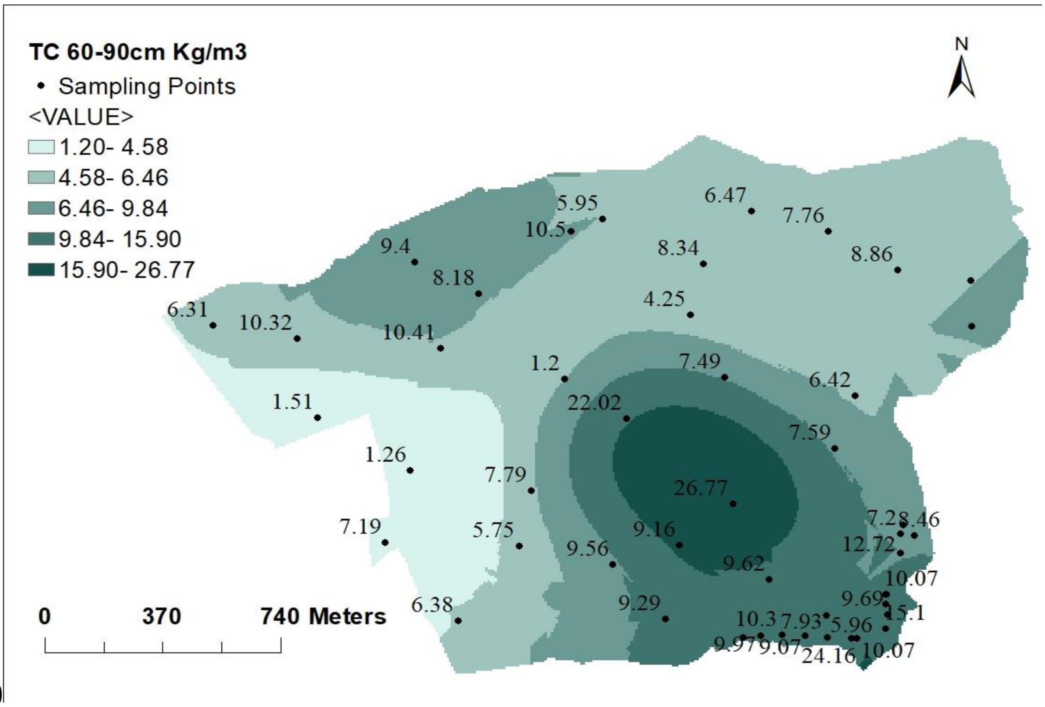
a)



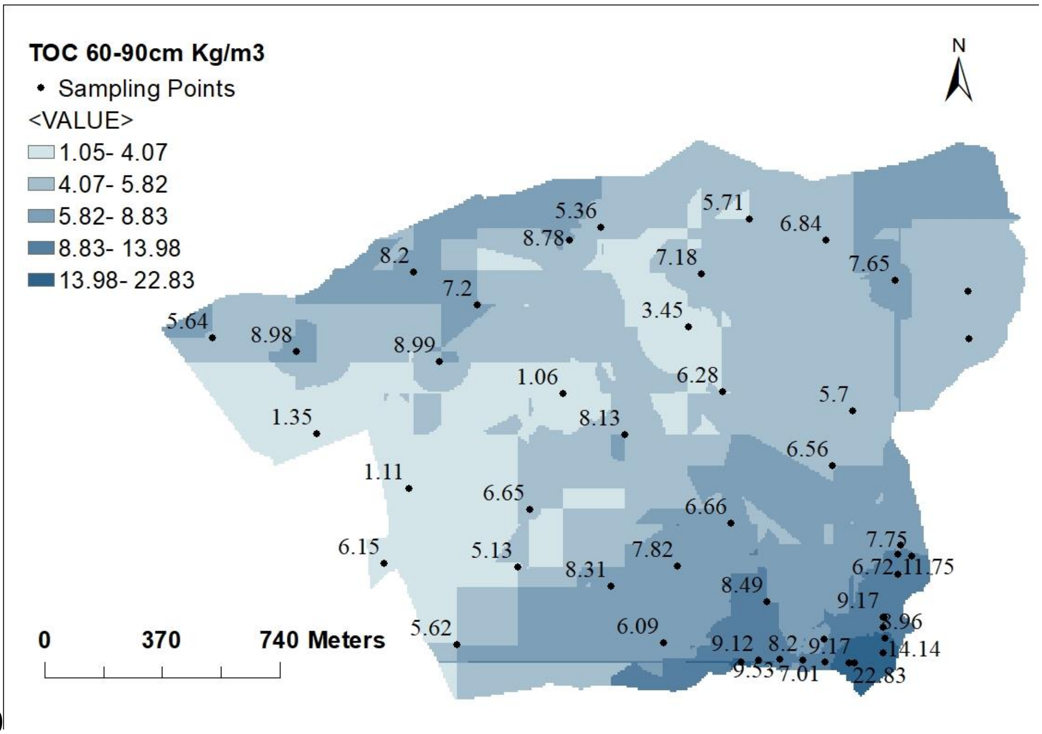
b)

Figure A. 5 Measured carbon and the carbon value estimated by Kriging for 30-60 cm soil at Nafferton Farm.

The labels aside the sampling plots are the measured carbon value; the classification of counters in the legend shows the estimated value ranges for soil carbon by Kriging. a) total carbon; b) organic carbon.



a)



b)

Figure A. 6 Measured carbon and the carbon value estimated by Kriging for 60-90 cm soil at Nafferton Farm.
The labels aside the sampling plots are the measured carbon value; the classification of counters in the legend shows the estimated value ranges for soil carbon by Kriging. a) total carbon; b) organic carbon. Blank label: no soil samples in this location.

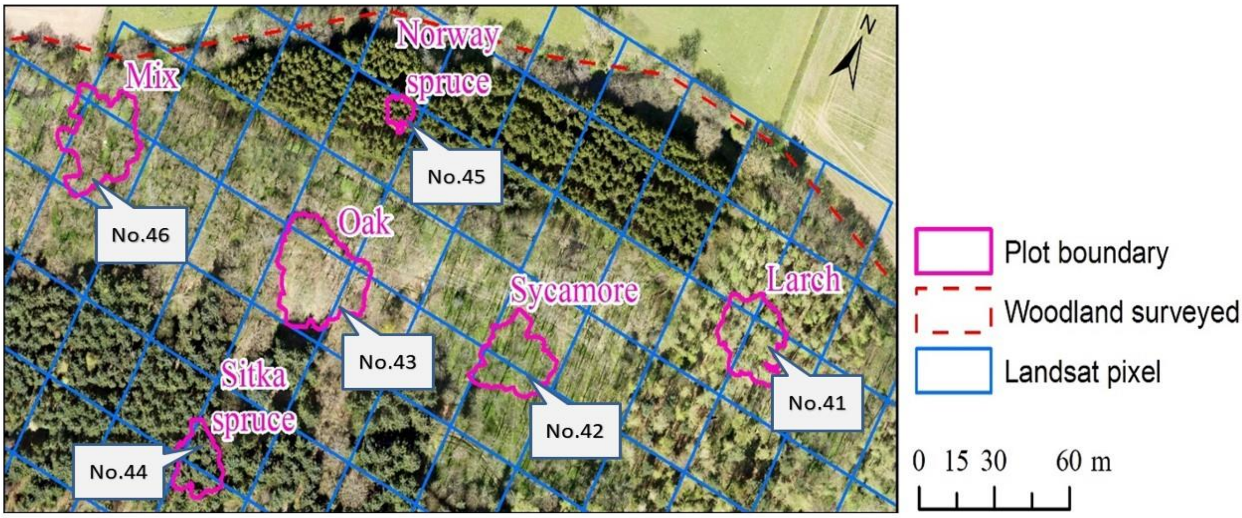


Figure A. 7 Aerial view of measured biomass areas at Cockle Park Farm woodland and the corresponding soil sampling IDs.
The reproduction of this figure has been approved by Berra (2018).

Appendix B

Appendix B1. Soil carbon and pH measurement and mineral characterization by X-ray diffraction over urban greenspace

At each sampling point, 0-30 cm soil cores were taken using a UMS gouge auger (diameter =25 mm) and stored in a 4 ℃ cold room. Soil samples were then dried at 105 ℃ for 24 h after removing the stones, roots, and leaves, and passed through a 4.75 mm sieve. Sample density was determined as the dry weight of soil core divided by its volume. Then, the soil subsample was ground into a fine powder using a ball mill (Tema Miller, SIEBTECHNIK GMBH, Mülheim an der Ruhr, Germany). For soil pH, 5 g ground samples were mixed with 12.5 mL deionised water, followed by stirring the solution well with a magnetic stirrer for 30 minutes. After settling the solution for one hour, soil solution pH was measured with a pH meter (900 multiparameter water quality meter, BANTE Instrument, Shanghai, China). The carbon mass content as % of homogenised soils was analysed using a LECO RC 612 (LECO Corporation; Saint Joseph, Michigan USA): organic carbon in soil was combusted to CO_2 between 150-450 ℃, and inorganic carbon would be released later when the furnace temperature rose to 1000 ℃. This method is preferable to acid treatment which may result in losses of organic forms by leaching before they are oxidised (Siavalas et al., 2013). Soil carbon content per square meter was calculated as below:

Soil carbon content of the 0-30 cm topsoil (Kg·m^{-2}) = Soil bulk density (Kg·m^{-3}) × Soil depth (0.3 m) × Carbon mass (%) from LECO *Equation B1*

Powder X-ray diffraction (XRD) was used to charactize the mineral compositions. XRD was performed on 10 selected, powdered soil samples from the urban campus and suburban sports area (5 samples each) with a Philips X'Pert – PRO theta-theta PW3050/60 diffractometer in Bragg-Brentano geometry, employing a Copper Line Focus X-ray tube and an X'Celerator 1D-detector (www.malvernpanalytical.com/). Soil samples were mounted on a Si zero-background holder to a PW3064 sample spinner. The data collection was carried out from 2θ=5-90° with a step of 0.0334°, and an acquisition time of per step was 10 s. The original data were corrected using CrystalDiffract (Version 6.9.0, http://crystalmaker.com/) and the XRD pattern of reference minerals such as calcite, kaolinite and quartz were downloaded from the CrystalMaker library (Version 10.4.3, http://crystalmaker.com/).

Appendix B2. Processing i-Tree Eco and calculations of the tree carbon density as well as tree cover

i-Tree Eco (https://www.itreetools.org/) requests users to define the location, population, and weather condition of the study area, and import the relative parameters of studied trees: DBH, height, species, crown diameter, tree age, the status of tree, street tree or not, etc. The more tree information users offer, the more precise results will be obtained, but the DBH and tree species are the two basically required factors in i-Tree Eco. Once importing the whole group of data and sending it to the i-Tree Eco database platform, users would receive a comprehensive formatted report summarising the composition and structure of trees in the study areas, including carbon storage and sequestration of trees, pollution removal performance, air quality health impacts, etc. Because it's an easily learned process and free download, this software has been applied in many cities (Rocco et al., 2018, Hand et al., 2019). Depending on the available data for the campus trees, this study input up to seven variables to i-Tree Eco: tree species, DBH, total height, crown base height, crown width (north-south; east-west), street tree or not, maintenance recommended. In addition, when applying tree biomass allometric regression equations, the raw DBH of trees was the fundamental parameter while the total height of trees was also needed for some tree species.

i-Tree Eco can report tree carbon including aboveground and belowground biomass together, but allometric biomass equations report only aboveground biomass initially. Therefore, when calculating tree biomass according to empirical equations, the total tree carbon stock can be calculated from the aboveground biomass as below:

Total tree carbon stock (Kg)= Aboveground biomass of total trees (Kg) × 1.28 × 0.5
Equation B2

1.28 is the conversion factor between aboveground biomass and total biomass of plants; 0.5 is the common conversion factor between plant biomass and carbon (Lal and Augustin, 2011). Carbon storage of per unit tree cover was calculated as:

Tree carbon storage (Kg· m^{-2}) = Tree carbon stock (Kg)÷ Tree cover (m^2) Equation B3

Tree cover shape was assumed to be an ellipse (Shaw, 2005) and its area would be obtained by:

Tree cover (m^2) = π× the major radius of the ellipse (m)× the minor radius of the ellipse (m)
Equation B4

	Category	The area of green space (hectares)	Ratio of each objective to the total urban green space in the city of Newcastle upon Tyne
Newcastle University	Four Sports Grounds	16	0.90%
	Central Campus	9	0.50%
Newcastle City [a]	Outdoor Sport (Pitches& Private) [b]	46	2.60%
	Churchyards and cemeteries	70	4.00%
	Allotment	75	4.30%
	Park and Recreation Ground [c]	200	11.30%
	Amenity Green Space [d]	226	12.70%
	Education	331	18.70%
	Accessible Natural Green Space [e]	825	46.50%
	Total green land soils	**1, 774**	**100.00%**

Table B. 1 Area of greenspace in Newcastle University's urban campus and the city of Newcastle upon Tyne, and the ratio of each land component to the total green space owned by Newcastle City.

[a]: Sources and the precise standards defining each land type can be found in Newcastle City Council (2018).
[b]: Outdoor Sport comprises the publicly accessible sports pitches such as rugby and bowling greens, as well as private outdoor space with the limited public access among the local communities.
[c]: Park and Recreation Ground comprises the general open space surrounding play areas, sports facilities, etc.
[d]: Amenity Green Space includes areas open to free which are predominantly covered with (mown) grass, less likely to mark the entrance, and normally without management for the specific function as a park.
[e]: This typology brings together natural and semi-natural green spaces consisting of meadows, woodland and copse, most of which present the similar natural characteristics and wildlife value.

Common Name	***Scientific Name***	**Biomass equation**	**Coefficients**	**Equation Sources**	**Number of tree occurrence**	**Percentage occurrence**
Apple	*Malus*	$AGB=a*[(DBH*0.394)^{2}*h*3.281)]^{b}*0.454$	a=0.30645, b=0.93397	(Clark, 1986)	2	0.4%
Ash	*Fraxinus excelsior*	$AGB=a+b*DBH+c*DBH^{2}$	a=1.6895, b=-1.942, c=0.6678	(Albert et al., 2014)	29	6.1%
Beech	*Fagus sylvatica*	$AGB=a*DBH^{b}*H^{c}$	a=0.0306, b=2.347, c=0.590	(Zianis et al., 2005)	4	0.8%
Broad-leaved Cockspur	*Crataegus prunifolia*	$Log_{10}(AGB)=[Log_{10}(a)+b*Log_{10}(DBH*0.394)]*0.454$	a=2.034, b=2.6349	(Brenneman et al., 1978)	1	0.2%
Callery Pear	*Pyrus calleryana "Chanticleer"*	$AGB=a*DBH^{b}$	a=0.129296875, b=2.310647	(Aguaron and McPherson, 2012)	2	0.4%
Cappadocian maple	*Acer cappadocicum*	$AGB=Exp(a+b*lnDBH)$	a=-1.9123, b=2.3651	(Jenkins et al., 2003)	2	0.4%
Cherry Plum	*Prunus cerasifera*	$AGB=a*DBH^{b}$	a=0.129296875, b=2.310647	(Aguaron and McPherson, 2012)	1	0.2%
Common Lime	*Tilia X europaea*	$AGB=Exp(a+b*lnDBH+c*lnh)$	a=-3.032, b=2.115, c=0.538	(Čihák et al., 2014)	4	0.8%
Common Oak	*Querous robur*	$AGB=DBH^{a}*h^{b}*exp(c)$	a= 2.00333, b= 0.85925, c= –2.86353	(Zianis et al., 2005)	4	0.8%
Copper Beech	*Fagus sylvatica "Purpurea"*	$AGB=a*(DBH)^{b}*h^{c}$	a=0.0523, b=2.12, c=0.655	(Wutzler et al., 2008)	7	1.5%
Crab Apple	*Malus sylvestris*	$AGB=a*DBH^{b}$	a=0.1293, b=2.310647	(Aguaron and McPherson, 2012)	2	0.4%
Dawn Redwood	*Metasequoia glyptostroboides*	$AGB=a*DBH^{b}$	a=0.0787, b=2.4086	(Williams et al., 2003)	8	1.7%

Deodar	*Cedrus deodara*	$AGB=a*(DBH^2*h)^b$	a=0.1779, b=0.8103	(Ali et al., 2016)	1	0.2%
Downy Birch	*Betula pubescens*	$AGB=a*(DBH*10)^b$	a=0.00019, b=2.0832	(Johansson, 2007)	4	0.8%
Elder	*Sambucus nigra*	AGB=Exp(a+b*lnDBH)	a=-2.4800, b=2.4835	(Jenkins et al., 2003)	1	0.2%
English Elm	*Ulmus procera*	$AGB=a*(DBH*0.394)^2*(h*3.281)^b*0.454$	a=0.19128, b=0.91936	(Clark, 1986)	4	0.8%
Field Maple	*Acer campestre*	$AGB=a*DBH^{(b+c*DBH)*}h^d$	a=0.2591， b=1.4186, c=0.0203, d=0.4730	(Albert et al., 2014)	3	0.8%
Goat Willow	*Salix caprea*	ln(AGB)=a+b*ln(DBH)	a=-2.4441, b=2.4561	(Chojnacky et al., 2014)	1	0.2%
Golden Weeping Willow	*Salix Sepulcralis Chrysocoma*	ln(AGB)=a+b*ln(DBH)	a=-2.4441, b=2.4561	(Chojnacky et al., 2014)	1	0.2%
Grey Alder	*Alnus incana*	$AGB=a*(DBH*10)^b$	a=0.000499, b=2.337592	(Johansson, 1999)	1	0.2%
Handkerchie f tree	*Davidia involucrata*	AGB=Exp(a+b*lnDBH)	a=-2.48, b=2.4835	(Jenkins et al., 2003)	1	0.2%
Hawthorn	*Crataegus mono gyna*	AGB=Exp(a+b*lnDBH)	a=-2.48, b=2.4835	(Jenkins et al., 2003)	14	3.0%
Hazel	*Corylus avellane*	$AGB=a*h*DBH^2+b$	a=0.0364, b=0.0308	(Albert et al., 2014)	1	0.2%
Highclere Castle	*llex x altaclerensis*	AGB=Exp(a+b*lnDBH)	a=-2.48, b=2.4835	(Jenkins et al., 2003)	6	1.3%
Himalayan birch	*Betula utilis*	AGB=a/(1+Exp(-(DBH-b)/c)	a=457.5, b=26.22, c=5.922	(Alam and Nizami, 2014)	8	1.7%
Holly	*llex aquifolium*	AGB=Exp(a+b*lnDBH)	a=-2.48, b=2.4835	(Jenkins et al., 2003)	4	0.8%
Holm Oak	*Quercus ilex*	$AGB=a+b*DBH^2*h$	a=-0.6165, b=0.03582	(Zianis et al., 2005)	2	0.4%
Hornbeam	*Carpinus betulus*	$AGB=a*(DBH)^b$	a=0.258, b=2.1748	(Suchomel et al., 2012)	2	0.4%

Horse Chestnut	*Aesculus hippocastanum*	AGB=Exp(a+b*lnDBH)	a=-2.4800, b=2.4835	(Jenkins et al., 2003)	2	0.4%
Irish Yew	*Taxus baccata Fastgiata*	AGB=Exp(a+b*lnDBH)	a=-2.4623, b=2.4852	(Chojnacky et al., 2014)	2	0.4%
Italian alder	*Alnus cordata*	AGB=Exp(a+b*lnDBH)	a=-2.2094, b=2.3867	(Jenkins et al., 2003)	1	0.2%
Kanzan Cherry	*Prunus serrulata*	ln(AGB)=a+b*ln(DBH)	a=-2.2118, b=2.4133	(Chojnacky et al., 2014)	15	3.2%
Laburnum	*Laburnum anagyroides*	$AGB=(a*DBH^b)/1000$	a=57.47, b=2.418	(M'Rabet et al., 2017)	1	0.2%
Large-leaved Lime	*Tilia platyphyllos*	AGB=Exp(a+b*lnDBH+c*lnh)	a=-3.032, b=2.115, c=0.538	(Čihák et al., 2014)	95	20.1%
Lawson Cypress	*Chamaecyparis lawsoniana*	AGB=Exp(a+b*lnDBH)	a=-1.9615, b=2.1063	(Chojnacky et al., 2014)	4	0.8%
Leyland Cypress	*X Cupressocyparis leylandii*	AGB=Exp(a+b*lnDBH)	a=-1.9615, b=2.1063	(Chojnacky et al., 2014)	4	0.8%
Lombardy Poplar	*Populus nigra "Italica"*	AGB=Exp(a+b*lnDBH)	a=-2.2094, b=2.3867	(Jenkins et al., 2003)	1	0.2%
London Plane	*Platanus × hispanica*	AGB=Exp(a+b*lnDBH)	a=-2.2118, b=2.5349	(Chojnacky et al., 2014)	8	1.7%
Maidenhair t ree	*Ginkgo biloba*	Log_{10} (AGB)=a+b*Log_{10}(DBH)	a=-1.561, b=2.439	(Son and Kim, 1998)	1	0.2%

Table B. 2 Summary of the total tree species and their corresponding common scientific name, allometric biomass equations with coefficients and reference sources, and the number of trees from every species.
The units of DBH and height from different original references might be different and some of the parameters therefore needed to be converted during the calculation. This table shows the DBH in centimetre and height (h) in meter.

	DBH (cm)					Height (m)				
	Young	Semi Mature	Early Mature	Mature	Mean	Young	Semi Mature	Early Mature	Mature	Mean
Large-leaved Lime	n.a	23.85	24.5	44.85	40.15	n.a	10	12	16.58	14.82
SD	n.a	9.608	8.96	17.58	17.98	n.a	3.16	1.6	2.48	3.74
Sycamore	n.a	n.a	40	50.67	50.28	n.a	n.a	14	14.73	14.7
SD	n.a	n.a	0	16	15.82	n.a	n.a	0	3.27	3.21
Rowan	10	13.14	17.2	48	16.71	10	7.21	8.33	9	8.06
SD	0	3.66	5.19	25.46	10.26	0	1.93	1.68	1.41	1.86
Ash	n.a	n.a	20.71	50.41	43.24	n.a	n.a	9.43	13.95	13.03
SD	n.a	n.a	3.45	21.2	22.51	n.a	n.a	1.9	3.18	3.68
Silver Birch	n.a	n.a	15.71	42.5	17.95	n.a	n.a	11.33	14	11.57
SD	n.a	n.a	1.79	3.54	8.12	n.a	n.a	0.97	0	1.2
Norway Maple	n.a	45	21	31.36	26.9	n.a	8	10.75	11.82	10.6
SD	n.a	0	12.74	11.2	12.34	n.a	0	2.12	1.89	1.73
Swedish Whitebeam		n.a	39.29	54	47.94	n.a	n.a	10	10.4	10.24
SD	n.a	n.a	11.7	17.76	16.87	n.a	n.a	2.24	2.07	2.08
Kanzan Cherry	n.a	45	29.5	46.25	35	n.a	8	10.4	13	10.93
SD	n.a	0	11.65	37.05	21.13	n.a	0	2.22	3.46	2.79

	Tree Cover Area (m^2)					Statistical significance as the change of maturity (p value)		
	Young	Semi Mature	Early Mature	Mature	Mean	DBH	Height	Tree cover area
Large-leaved Lime	n.a	41.81	55.61	68.13	63.21	0.107	**<0.001**	**0.017**
SD	n.a	11.14	43.63	31.55	32.22			
Sycamore	n.a	n.a	78.54	97.5	96.79	0.077	0.351	0.556
SD	n.a	n.a	0	44.86	44.15			
Rowan	15.45	22.78	27.28	61.26	26.38	**<0.001**	0.333	**<0.001**
SD	7.26	9.8	10.6	24.44	14.04			
Ash	n.a	n.a	31.98	88.39	74.78	**0.001**	**<0.001**	**0.037**
SD	n.a	n.a	11.74	66.97	63.22			
Silver Birch	n.a	n.a	24.24	56.94	27.08	**<0.001**	**<0.001**	**<0.001**
SD	n.a	n.a	5.4	9.44	10.92			
Norway Maple	n.a	113.1	42.02	64.55	57.96	0.18	0.287	**0.024**
SD	n.a	0	26.21	22.93	28.67			
Swedish Whitebeam	n.a	n.a	72.14	75.4	74.06	0.307	0.052	0.829
SD	n.a	n.a	41.26	19.08	29.08			
Kanzan Cherry	n.a	94.25	76.65	184.57	106.6	0.455	0.55	0.353
SD	n.a	0	42.34	231.29	122.49			

Table B. 3 Summary of the eight most frequently occurring trees and their corresponding mean diameter at breast height (DBH, cm), height (m), canopy cover (m^2) within the different age classifications and the statistical significance caused by the division of age groups.
SD: Standard deviation. Significant (p<0.05) findings have been highlighted; n.a: no data for that group.

		STC	SOC	SIC
Soil pH	Coefficient	0.381	0.296	0.605
	p value	**0.013**	0.057	<**0.001**

Table B. 4 Pearson correlation between soil pH and soil carbon content.
STC: soil total carbon; SOC: soil organic carbon; SIC: soil inorganic carbon. Correlation is significant at the 0.05 level.

Parameter	Unit	Data
Greenspace area in Newcastle University city campus	hectares	25.0
Total current terrestrial carbon stock (soils+trees)	tonnes	4383
Carbon storage of Ash	tonnes per hectare canopy	146.7
Carbon storage of Swedish Whitebeam	tonnes per hectare canopy	135.7
Carbon storage of Sycamore	tonnes per hectare canopy	83.6
Carbon storage of Large Leaved Lime	tonnes per hectare canopy	72.1
Mean tree carbon storage based on top 4 tree species	tonnes per hectare canopy	109.5
Woodland soil carbon storage	tonnes/ha	230.5
Terrestrial carbon storage of woodland (soil + 4 top tree species)	tonnes/ha	340.0
Total future terrestrial carbon stock (soil + 4 top tree species)	tonnes	8500.6
The average age of Ash, Swedish Whitebeam, Large-leaved Lime, and Sycamore	years	57
Estimated annual carbon sequestration	tonnes/year	72.2
Carbon emission of Newcastle University in 2019/20	tonnes	6406
The percentage of estimated annual carbon storage to the current annual carbon emission	%	1.13

Table B. 5 Scenario 1-Campus green area does not change but C storage increases by converting all campus green areas to the mixed woodland by 2030 with 4 top species showing the highest C storage ability.

Parameter	Unit	Data
Greenspace area Newcastle City	hectares	1774
Total current terrestrial carbon stock (soils + trees)	tonnes	422241
Carbon storage of Ash	tonnes per hectare canopy	146.7
Carbon storage of Swedish Whitebeam	tonnes per hectare canopy	135.7
Carbon storage of Sycamore	tonnes per hectare canopy	83.6
Carbon storage of Large Leaved Lime	tonnes per hectare canopy	72.1
Mean tree carbon storage based on top 4 tree species	tonnes per hectare canopy	109.5
Woodland soil carbon storage	tonnes/ha	230.5
Terrestrial carbon storage woodland (soil + 4 top tree species)	tonnes/ha	340.0
Total future terrestrial carbon stock (soil + 4 tree species)	tonnes	603204
The average age of Ash, Swedish Whitebeam, Lime, and Sycamore	years	57
Carbon emission of Newcastle City in 2018	tonnes	335400
Estimated annual carbon sequestration	tonnes/year	3174.8
The percentage of estimated annual carbon storage to the current annual carbon emission		0.95%

Table B. 6 Scenario 2- All green areas of Newcastle city are converted to the woodland with 4 top species showing the highest C storage ability.

Parameter	Unit	Data
Greenspace area in Newcastle City	hectares	1774
Total current terrestrial carbon stock (soils + trees)	tonnes	422241
Carbon storage of Ash	tonnes per hectare canopy	146.7
Carbon storage of Swedish Whitebeam	tonnes per hectare canopy	135.7
Carbon storage of Sycamore	tonnes per hectare canopy	83.6
Carbon storage of Large-leaved Lime	tonnes per hectare canopy	72.1
Mean tree carbon storage based on top 4 tree species	tonnes per hectare canopy	109.5
Woodland soil carbon storage	tonnes/ha	230.5
Terrestrial carbon storage of woodland (soil + 4 top tree species)	tonnes/ha	340.0
New woodland area	hectares	33.7
Total future terrestrial carbon stock (soil + 4 top tree species)	tonnes	426064
The average age of Ash, Swedish Whitebeam, Large-leaved Lime, and Sycamore	years	57
Carbon emission of Newcastle City in 2018	tonnes	335400
Estimated annual carbon sequestration	tonnes/year	67.07
The percentage of estimated annual carbon storage to the current annual carbon emission	%	0.020

Table B. 7 Scenario 3 – Tree coverage in city green area increases to its full potential by 2050 by introducing mixed woodland with 4 top species showing the highest C storage ability.

Parameter	Unit	Data
Greenspace area in Newcastle City	hectares	1774
Total current terrestrial carbon stock (soils+trees)	tonnes	422241
Mean tree carbon storage	tonnes per hectare canopy	76.6
Woodland soil carbon storage	tonnes/ha	230.5
Terrestrial carbon storage woodland (soils+trees)	tonnes/ha	307.1
New woodland area	hectares	33.7
Total future terrestrial carbon stock (soils+trees)	tonnes	424954
The average tree age	years	57
Carbon emission of Newcastle City in 2018	tonnes	335400
Estimated annual carbon sequestration	tonnes/year	47.60
The percentage of estimated annual carbon storage to the current annual carbon emission	%	0.014

Table B. 8 Scenario 4 – Tree coverage in city green area increases to its full potential by 2050 and C storage ability of woodlands remains unchanged (the mean value as measured in this study).


a)
Campus_lawn soil total carbon (Kg/m2)
Sampling Points
Road
Building
19.14
28.27
18.5
26.4
32.1
14.85
21.26
27.78
24.18
32.63
17.73
13.88
N
0 25 50 100 Meters
b)
Campus_lawn soil organic carbon (Kg/m2)
Sampling Points
Road
Building
18.98
11.92
10.94
8.45
15.72
13.67
13.78
20.31
19.47
20.21
20.38
13.21
N
0 25 50 100 Meters

c)

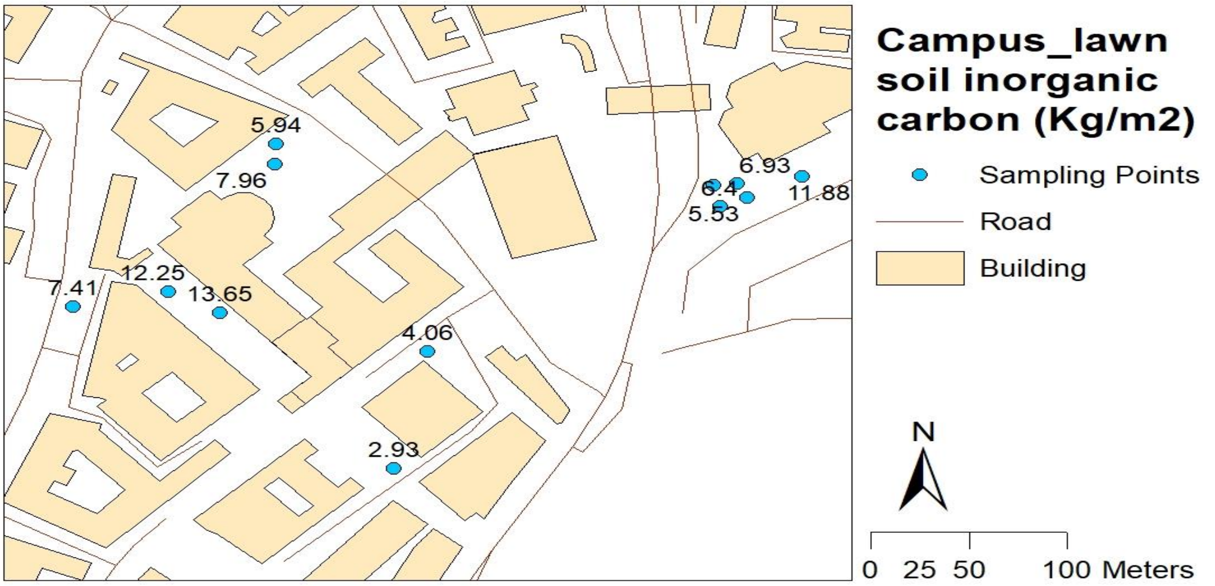
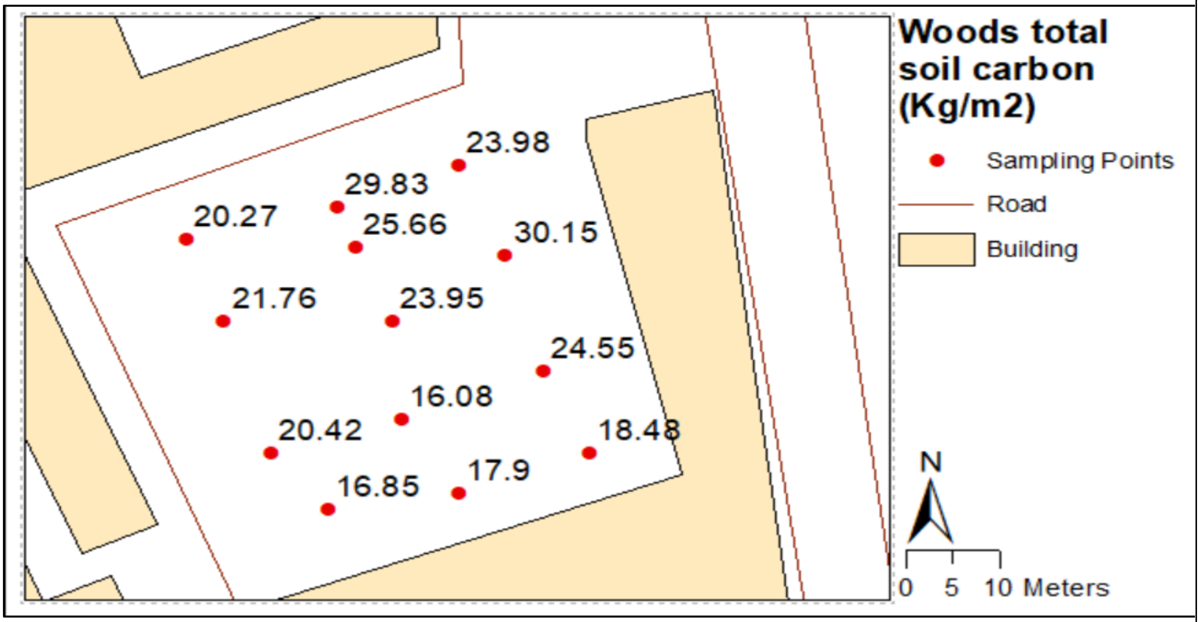
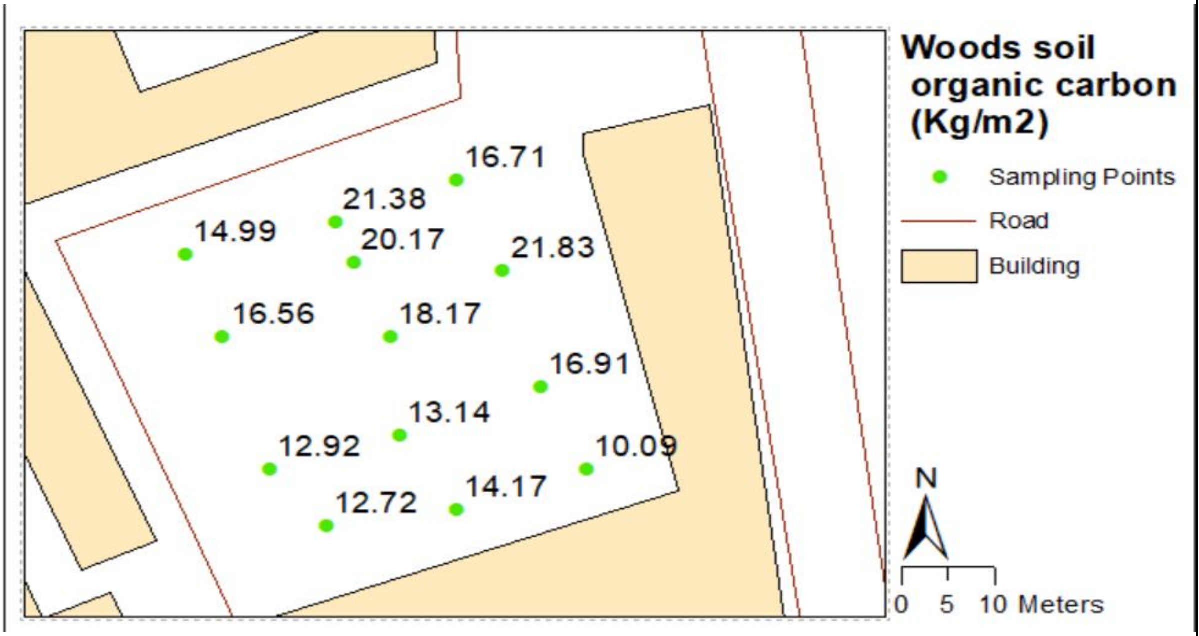


Figure B. 1. 0-30 cm soil carbon storage (Kg·m^{-2}) from 12 sampling points across lawns of the central campus of Newcastle University.
a): total carbon storage of soils; b) organic carbon storage of soils; c) inorganic carbon storage of soils.

a)


Woods total soil carbon (Kg/m2)
Sampling Points
Road
Building
23.98
29.83
20.27
25.66
30.15
21.76
23.95
24.55
16.08
20.42
18.48
17.9
16.85
N
0 5 10 Meters

b)


Woods soil organic carbon (Kg/m2)
Sampling Points
Road
Building
16.71
21.38
14.99
20.17
21.83
16.56
18.17
16.91
13.14
12.92
10.09
14.17
12.72
N
0 5 10 Meters

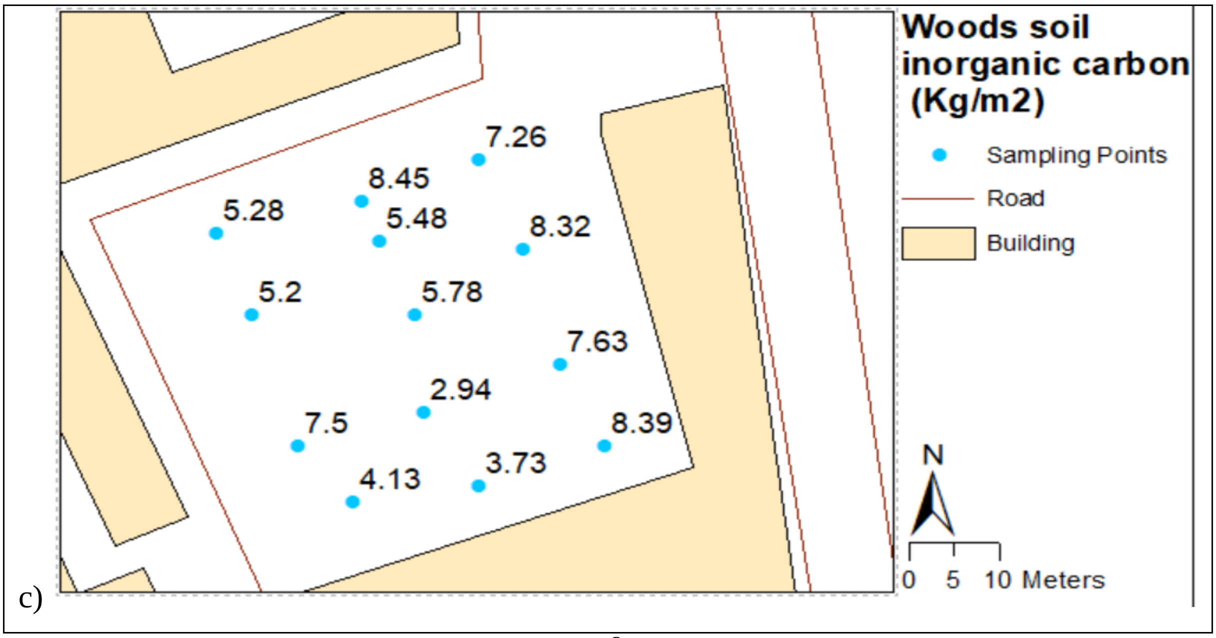
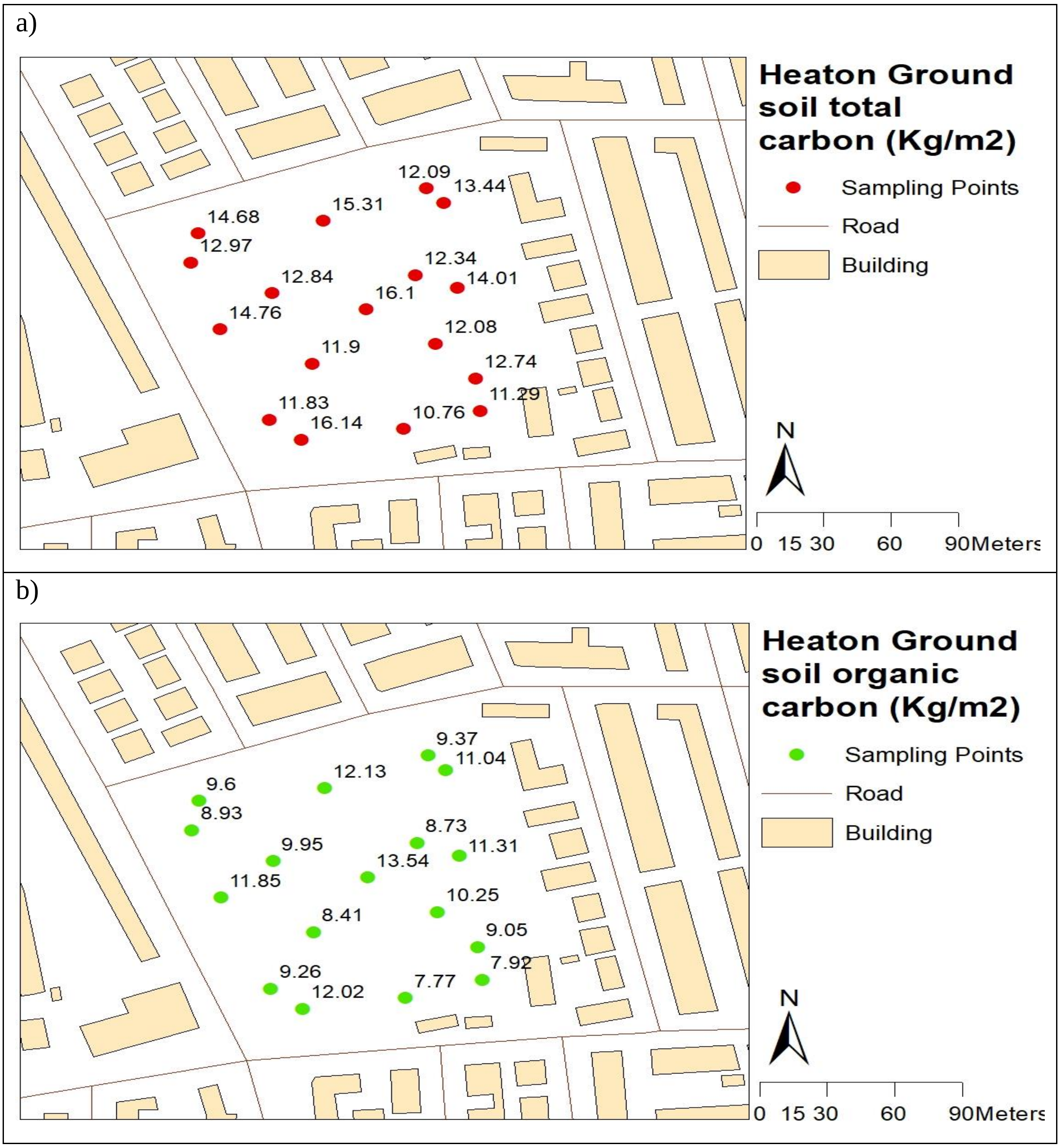


Figure B. 2 0-30 cm soil carbon storage (Kg·m^{-2}) from 13 sampling points across a woodland in the central campus of Newcastle University.
a): total carbon storage of soils; b) organic carbon storage of soils; c) inorganic carbon storage of soils.


a)
Heaton Ground soil total carbon (Kg/m2)
Sampling Points
Road
Building
12.09
13.44
15.31
14.68
12.97
12.34
12.84
14.01
16.1
14.76
12.08
11.9
12.74
11.29
11.83
10.76
16.14
N
0 15 30 60 90Meters
b)
Heaton Ground soil organic carbon (Kg/m2)
Sampling Points
Road
Building
9.37
11.04
12.13
9.6
8.93
8.73
9.95
11.31
13.54
11.85
10.25
8.41
9.05
7.92
9.26
7.77
12.02
N
0 15 30 60 90Meters

c)

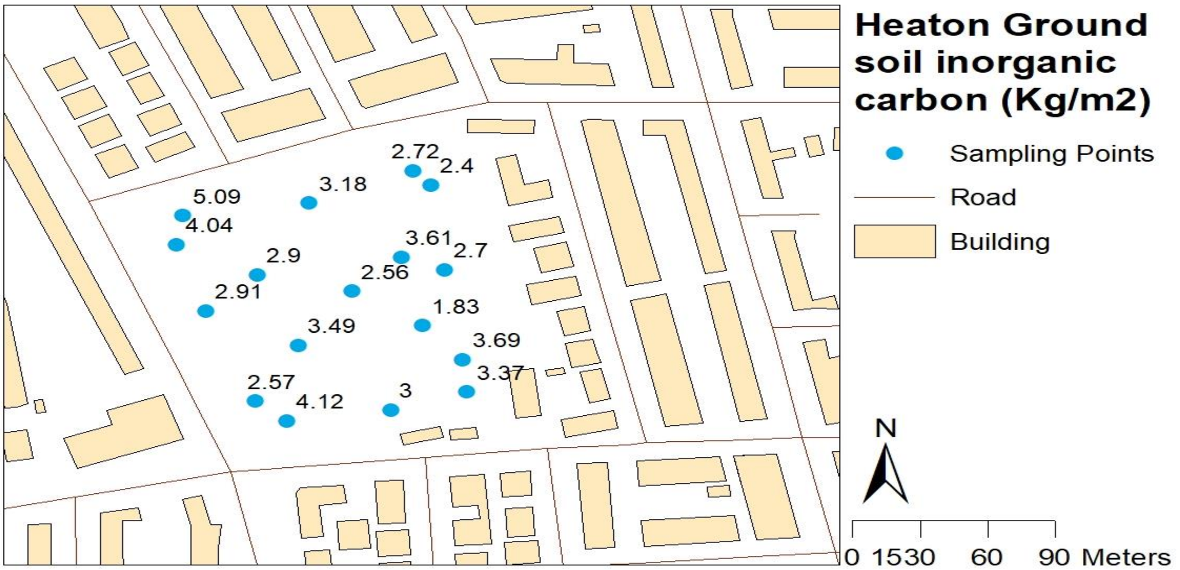


Figure B. 3 0-30 cm soil carbon storage (Kg·m^{-2}) from 17 sampling points across the Heaton Sports Ground of Newcastle University.

a): total carbon storage of soils; b) organic carbon storage of soils; c) inorganic carbon storage of soils.

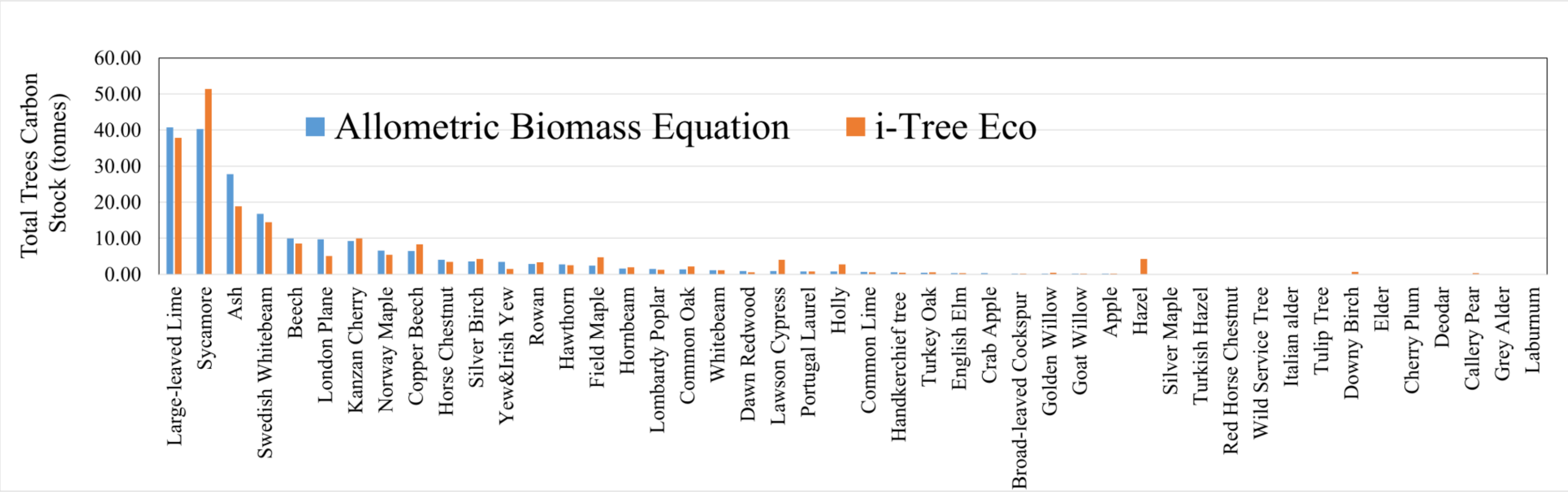


Figure B. 4. Total carbon stock (tonnes) of trees from 46 tree species on the campus of Newcastle University calculated by i-Tree Eco and allometric biomass equations, respectively.

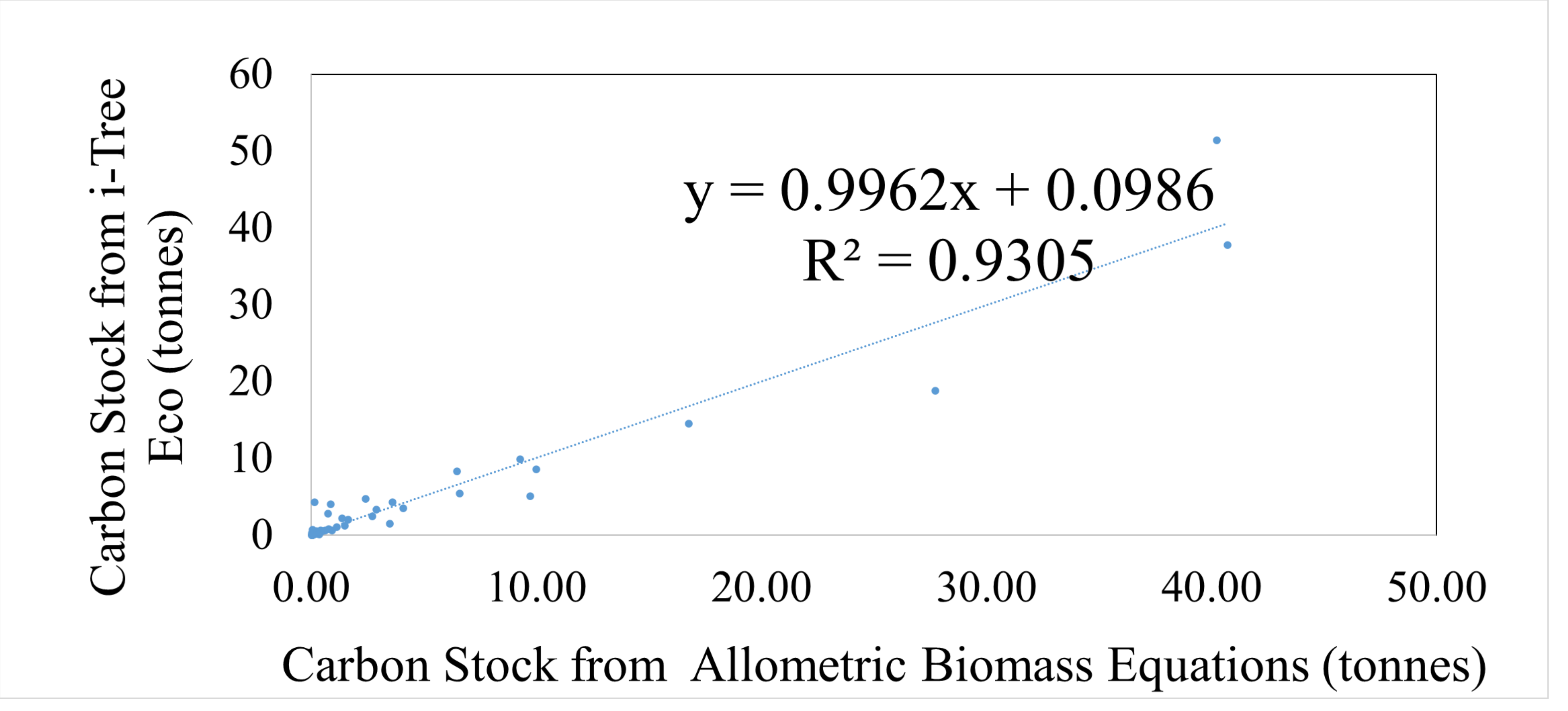


Figure B. 5 Linear relationship between allometric biomass equations and i-Tree Eco for the total carbon stock of trees from each species (46 species in total).

Appendix C

- **Application of wheat straw biochar and wheat straw pellets in 2 lysimeters**

This study wanted to apply an equivalent amount of carbon weight either as biochar or wheat straw to two lysimeters. Biochar would be uniformly mixed with sandy loam soil at a rate of 2% (w/w) corresponding to an application rate of 48 tonnes·ha^{-1}, assuming a soil bulk density of 1,600 kg·m^{-3} and soil depth of 0.15 m (Obia et al., 2016, Gamage et al., 2016), therefore this experiment required 43.2 kg to amend a total volume of 1.35 m^{3} of sandy loam soil, as per soil depth of 0.15 m within lysimeter BC (Table C.1). The weight of total carbon in wheat straw biochar is 690.4 g·kg^{-1} and the weight of total carbon in wheat straw is 463.3 g·kg^{-1} (UK Biochar Research Centre, 2014). Therefore, for lysimeter WP, if we apply biochar at a rate of 48 tonnes·ha^{-1}, we will need to apply wheat straw at a rate of 71 tonnes·ha^{-1} to obtain an equivalent carbon weight balance. We will therefore need 64.35 kg of wheat straw to be amended in the top 0.15 m layer of soil in Lysimeter WP (Table C.1).

Item	Value
Soil depth applying the soil amendments(m)	0.15
Soil bulk density (kg/m3)	1600.00
Surface area of lysimeter (m2)	9.00
Soil volume applying the soil amendments (m3)	1.35
Soil weight applying the soil amendments (kg)	2160.00
The rate applying biochar to soils (weight %)	0.02
Applied biochar weight (kg)	**43.20**
The weight% of total carbon in biochar (g/kg)	690.40
Carbon from biochar added into the lysimeter BC (g)	29825.28
The weight% of total carbon in wheat straw pellets (g/kg)	463.30
Carbon from wheat straw pellets added into the lysimeter BC (g)	29825.28
Applied wheat straw pellets weight (kg)	**64.38**
Rate to apply wheaT straw pellets into the lysimeter WP (weIght %)	71.53

Table C. 1 Calculation steps for how to apply the different rates of biochar and wheat straw pellets with the equivalent amount of carbon into two lysimeters.

- **Environmental sensors in the lysimeters**

Before adding amendments, a total of 12 sensors (METER GROUP ECH2O 5TE SOIL MOISTURE SENSOR; Weiz, Austria) were installed in both lysimeters (six for each) to monitor the soil data every 15 minutes. Every sensor has three probes to record soil temperature, soil volumetric water content and bulk electrical conductivity, respectively. First in the right side of one lysimeter, three sensors were installed at the same point but in three soil interval layers (10 cm, 50 cm, 80 cm). Likewise, horizontally paralleling with the point at the right side, another three sensors were put at the left side of the lysimeter still at three soil depths, respectively.

- **Soil preparation and carbon calculation**

Soil samples collected *in situ* were sealed in polyethylene bags and stored in the cold room (4 °C) for further processing. After removing the coarse debris (grass roots, stones, etc), soil samples were dried at 105 °C in an oven for 48 hrs until the weight remained constant, and afterwards passed through a 4.75 mm sieve. Later, soil was ground to a fine powder for two minutes with a laboratory disc mill (TEMA Machinery Ltd, UK). The dry bulk density of the soil samples was determined as the dry weight of the samples divided by the soil core volume. The total soil carbon (TC, w/w %) and organic soil carbon content (TOC, w/w%) was measured by a LECO-RC 612 machine (LECO Corporation, Saint Joseph, Michigan, USA). The combustion process decomposed organic carbon to CO_2 between 150°C to 450°C, and inorganic carbon thereafter, below 1000°C. Thus the percentage of organic carbon and inorganic carbon of soils could be obtained by heating the soils in an oxidizing environment (LECO, 2018). The carbon storage in the topsoil layer was calculated by the mean C content multiplying with the soil bulk density:

$$\textit{Carbon storage } (Kg\cdot m^{-2}) = \textit{Carbon mass content } (w/w\ \%) \times \textit{Soil bulk density } (Kg\cdot m^{-3}) \times \textit{Top soil core depth } (0.15\ m) \qquad \textit{Equation C1}$$

The change of soil carbon contents achieved with the amendments was assessed in comparison with the initial soil carbon measured in the lysimeters before the soil amendment in June 2018.

- **Soil carbon isotope analysis**

Isotope analysis of selected soil samples and the raw soil amendments was performed by Iso-Analytical Limited (Crewe, United Kingdom). Stable isotope analysis results are shown as per mill (‰) units:

$$\delta\ Element(‰) = \frac{R_{Sample} - R_{Standard}}{R_{Standard}} \times 1000 \qquad \textit{Equation C2}$$

$R_{Standard}$ is the established isotope reference in terms of the standard Vienna Pee Dee Belemnite (PDB) and R_{Sample} is the molar ratio of light isotope such as $^{13}C/^{12}C$ or $^{18}O/^{16}O$. The reference material used during carbonate isotope analysis of samples in this study was IA-R022 (Iso-Analytical working standard calcium carbonate, $\delta^{13}C_{V\text{-}PDB}$ = -28.63 ‰ and $\delta^{18}O_{V\text{-}PDB}$ = -22.69 ‰). For measuring the carbon ($\delta^{13}C_{carb}$) and oxygen isotopes ($\delta^{18}O_{carb}$) of carbonate in soil, all samples, packed in Exetainer™ tubes, were reacted with phosphoric acid after flushing with 99% helium in a heated environment for 60 minutes. The samples in the tubes were left overnight to ensure all carbonate in soils had been converted to CO_2. Then, gaseous CO_2 was sampled from the tubes and analyzed by Continuous Flow-Isotope Ratio Mass Spectrometry (CF-IRMS) in a continuous He stream. Secondly, for analyzing carbon isotopes of organic matter ($\delta^{13}C_{org}$), subsamples of soil and soil amendments were acidified with hydrochloric acid. Afterwards, the samples were put at 60 °C in the oven for 2 hours, and then left outside the oven for the whole day. This is for liberating carbonate to CO_2. Then, after the centrifugation and disposing the acid, the samples were washed twice with distilled water, and centrifugated again. The treated soils were then oven dried at 60 °C and ground. Elemental Analyser - Isotope Ratio Mass Spectrometry (EA-IRMS) was used for measuring $\delta^{13}C_{org}$ where samples were combusted in an oxygen rich environment. The reference material used during $\delta^{13}C$ analysis of organic matter was IA-R001 (wheat flour, $\delta^{13}C_{V\text{-}PDB}$ = -26.43 ‰). This study used the isotope results of three standard materials: concrete, limestone and sand, which would be combined with the lysimeter soils to analyze the carbonate feature in the biochar/biomass amended soils. Among these three geological materials, Barrasford limestone is the local “Carboniferous limestone” of Newcastle; sand comes from “Permian Sand”, which is used in construction; concrete was collected from the demolition waste at Cockle Park Farm managed by Newcastle University.

In each lysimeter, 21 topsoil samples (five from 2018; eight from 2019; four from 2020/2021, respectively) and 13 subsurface soil samples (three from 2018; two from 2019; four from 2020/2021, respectively) were selected for the isotope analysis. The organic carbon isotope ratio ($\delta^{13}C_{org}$) of the two soil amendments (wheat straw pellets and wheat straw pellet biochar) and 34 soil samples from two lysimeters were determined by an Elemental Analyser - Isotope Ratio Mass Spectrometry (EA-IRMS); then, all soil samples were analysed by Continuous Flow-Isotope Ratio Mass Spectrometry (CF-IRMS) for the carbon and oxygen isotopes compositions of carbonate ($\delta^{13}C_{carb}$, $\delta^{18}O_{carb}$). To calculate the faction of carbon input (X) in the soils from the amendment sources, the following formula is applied here (Christensen et al., 2011):

$$X = \frac{\delta^{13}C\_(soil\_sample) - \delta^{13}\ C\_(soil\_initial)}{\delta^{13}C\_(amendment\_source) - \delta^{13}\ C\_(soil\_initial)} \qquad \textit{Equation C3}$$

The $\delta^{13}C_{soil\_sample}$ is the $\delta^{13}C$ of analysed soil samples being incubated; $\delta^{13}C_{soil\_initial}$ is the $\delta^{13}C$ of the soil sampled before adding soil amendments; $\delta^{13}C_{amendment\_source}$ is the $\delta^{13}C$ of wheat straw or biochar.

- **Soil infiltration test**

This study carried out soil infiltration tests monthly from February to June 2021. A double-ring infiltrometer test (Figure C.1) was prepared for the field infiltration measurement. As the image below shows, the small ring is 13.5 cm in height and 10 cm in diameter; the large ring is 20.5 cm in height and is 20 cm in diameter. The small ring was placed at the centre inside the large ring and both rings were hammered into the ground until 8 cm was left above the ground. Initially, the waterline in both rings should start from the same level, and the ring walls should be vertical. Timing with a stopwatch was started when the small ring was first filled with tab water, and then another stopwatch was started when the space between the small and large ring was filled with tab water. This step was done quickly. Changes in the water level were recorded at intervals of every minute in the inner ring, and it was also noted how much time it took for the water level to drop by a centimetre in the space between the two rings. In each lysimeter, the infiltration test was conducted twice. For the double-ring infiltrometer, infiltration rate is usually calculated only from the inner small ring (Johnson, 1963):

$$Infiltration\ Rate\ of\ the\ Inner\ Ring\ (cm \cdot min^{-1}) = \frac{The\ difference\ of\ the\ measured\ water\ levels\ between\ two\ readings\ (cm)}{The\ time\ spent\ between\ two\ readings\ (min)} \qquad \textit{Equation C4}$$

Results of the infiltration tests can be found in Figure C. 5.

- **Vegetation biomass carbon calculation**

The fresh weight of the cut biomass was recorded using a portable luggage scale on the site, and then a subsample of known weight (approx. 1Kg) was retained for laboratory analysis. The rest of the clippings was returned to the surface of each lysimeter. The biomass subsamples were dried to a constant weight at a 60 ℃ in the oven for 48 h, and a conversion value of 0.475 was used in this study to convert the dry biomass into carbon weight (Reed and Magnussen, 2004). Therefore, the carbon content of the above ground vegetation in the lysimeters could be calculated as:

$$Carbon\ in\ above\ ground\ vegetation\ (Kg) = Wet\ weight\ cuttings\ (Kg) \times \frac{Dry\ weight\ subsample\ (Kg)}{Wet\ weight\ subsample\ (Kg)} \times 0.475 \qquad \textit{Equation C5}$$

- **Leaching carbon calculation**

Daily rainfall records from World Weather Online (2018-2020) were used for estimating the rainfall volume passing through the lysimeter between the leachate collection days. For this calculation, the amount of water lost by evaporation was estimated as 0.05 $mm \cdot day^{-1}$ in winter, 0.2 $mm \cdot day^{-1}$ in spring and autumn, and 0.3 $mm \cdot day^{-1}$ in summer (Prudhomme and Williamson, 2013). Consequently, the estimated leachate volume was:

The total volume of water leaching through one lysimeter (m^3) = [Rainfall per day (mm) – Evaporation per day(mm)] ×10^{-3} × the number of days between two leachate collection dates × the surface area of the lysimeter (m^2) *Equation C6*

Then, this value would be multiplied with the leachate carbon concentration determined by the TOC analyser, which would be obtained as the sum of monthly carbon mass loss from the leachate between two consecutive samples collections, as per the equation below:

The monthly C lost with the leachate (Kg) = The total volume of water leaching through one lysimeter in a month (m^3) (data from Equation 6) × the dissolved total carbon concentration ($mg \cdot L^{-1}$) ×10^{-3} *Equation C7*

- **Leachate pH and carbonate alkalinity measurement**

pH of the leachate was measured in duplicate by a pH meter (JENWAY 3510, Dunmow, Essex, UK). Alkalinity was measured through titration (Lossie and Putz, 2009) in units of $CaCO_3$ $mg \cdot L^{-1}$, where 20 mL leachate was first filtered with a 0.45 um membrane (VWR

International, UK) and then put in a beaker on a magnetic stirrer which should work continuously during the titration. For the titration, a 0.1N H_2SO_4 solution was prepared. The initial pH of the filtered leachate was recorded when a stable reading was obtained, and then the leachate was titrated with 0.1N H_2SO_4 to the end point at pH=5, and the volume of H_2SO_4 used in total was recorded. Each leachate was titrated in duplicate as well. Then alkalinity was calculated using the equation below (Lossie and Putz, 2009):

Alkalinity (total inorganic carbonate) (mg $CaCO_3 \cdot L^{-1}$) = Volume of 0.1N H_2SO_4 used during the titration (mL)× 250 *Equation C8*

- **CO_2-C emission collection and measurement**
  - **1st method: Manually installed tubes to collect soil gas for CO_2 measurement in the lysimeter**

Generally, gas samples were collected between 9 am to 11 am on the sampling day from both lysimeters. On the lysimeter site, a new 10 mL syringe was inserted at one side of a 10 mL serum glass vial sealed by a rubber septum cap, and used to extract the original gas in the vial (Figure C. 2). Then, soil gas was withdrawn from the stainless tubes (shown in Figure 5. 1), and with another 10 mL syringe via the septum injected into the 10 mL serum glass vial (Figure C. 2). The gas from each individual buried tube was collected in triplicate. CO_2 emissions sampling ended in March 2020 because of the national lockdown during the COVID-19 pandemic. Therefore, from August 2018 to February 2020, a total of 384 soil gas samples from the buried tubes and 192 atmospheric samples were collected. All samples were processed by gas chromatography-mass spectrometry to give the CO_2 concentration as $g \cdot m^{-3}$, so it was necessary to convert the CO_2 concentration unit to the gas flux of CO_2-C ($g \cdot m^{-2} \cdot h^{-1}$). The equation for this conversion is given here:

$$CO_2 - C_{Flux} = \frac{12}{44} \times \frac{D_e \times (C_{air} - C_{soil})}{z}$$ *Equation C9*

D_e is effective diffusion coefficient of CO_2 in the topsoil ($m^2 \cdot h^{-1}$), C_{air} is the concentration of CO_2 in the air above soil surface ($g \cdot m^{-3}$), C_{soil} is the concentration of CO_2 ($g \cdot m^{-3}$) in the gas samples collected from a particular soil depth, z, which is also the specific length of the stainless steel tube (e.g. 0.01 m for our calculation); D_e can be obtained as per below:

$$D_e = \frac{D_m \times \varepsilon^{2.5}}{\Phi}$$ *Equation C10*

D_m is molecular diffusion coefficient of CO_2 in air ($m^2 \cdot h^{-1}$), ε is the soil air-filled porosity ($m^3 \cdot m^{-3}$), Φ is the total soil porosity ($m^3 \cdot m^{-3}$) (Werner et al., 2004). D_m can be estimated as:

$$D_m = (\frac{P_o}{P}) \times D_a \times \left(\frac{T_s}{T_o}\right)^{1.75} \qquad \textit{Equation C11}$$

P_o is the pressure under standard condition (Pa); P is the pressure (Pa) measured at the day collecting samples, and in this study P is assumed to equal with P_o; D_a is the molecular diffusion coefficient value under standard conditions ($m^2 \cdot h^{-1}$): 5.004×10^{-2} $m^2 \cdot h^{-1}$ for CO_2 at standard temperature and pressure (T_0= 273.15 K; P_0 = 101.3 kPa) (Salmawati et al., 2017). T_s is the soil temperature (Kelvin). The total soil porosity (Φ, $m^3 \cdot m^{-3}$) is calculated as:

$$\Phi = 1 - \frac{\rho_d}{\rho_P} \qquad \textit{Equation C12}$$

where ρ_d is the bulk soil density in this study (1,113 $Kg \cdot m^{-3}$) and ρ_p is the density of mineral soil (2,560 $Kg \cdot m^{-3}$) (Salmawati et al., 2017). The value of the soil air-filled porosity (ε, $m^3 \cdot m^{-3}$) is obtained by:

$$\varepsilon = \Phi - \mu \qquad \textit{Equation C13}$$

μ is the water-filled porosity ($m^3 \cdot m^{-3}$), which in this study equals the volumetric water content recorded by the sensors in the soils.

Finally, the monthly CO_2-C (Kg) lost via gas exchange with the atmosphere can be obtained as below:

The monthly CO_2-C lost via gas exchange with the atmosphere (Kg) = CO_2-C flux ($g \cdot m^{-2} \cdot h^{-1}$) × total hours per month (assume 30 days every month; 720 hours totally) × the surface area of the lysimeter (9 m^2)×10^{-3} *Equation C14*

- **2nd method: Using a portable CO_2 gas analyser**

The EGM-4 Environmental GAS Monitor for CO_2 (PP Systems, Amesbury, USA) was used monthly from March 2019 to February 2020. The operator's manual about this instrument can be found at: https://guidessimo.com/document/1361560/pp-systems-egm-4-operator-s-manual-54.html. The EGM-4 CO_2 gas analyzer is a menu driven instrument using infrared gas analysis technology and has a static chamber (diameter :10 cm, height: 16 cm, Figure C. 3) to detect the CO_2 gas emanating from the soil. After turning on the machine, the system will be in the warming process over 5 minutes and afterwards the screen will show the accomplishment of the desired temperature. Then, the static chamber was put onto the earth making sure the chamber is vertical, and inserting the chamber 2-3 cm into the soil. Following the instructions on the menu screen the measurements were started, where every single measurement took 124 seconds. At the start of the measurement, one measurement value displayed on the screen was the initial ppm of CO_2 gas in the chamber ($umol \cdot mol^{-1}$); during the measuring process, the CO_2 gas ppm value changed over time. The CO_2 gas ppm value in the chamber was recorded at the beginning and end. In each lysimeter, four locations were chosen on every sampling day to perform such measurements. The change of CO_2 gas ppm values over the measuring process is:

Δ CO_2 ppm over 124 seconds ($umol \cdot mol^{-1}$) = the initial value of CO_2 ppm ($umol \cdot mol^{-1}$)- the final value of CO_2 ppm ($umol \cdot mol^{-1}$) *Equation C15*

The rate of change of the CO_2 ppm value is calculated as:

Δ CO_2 ppm per second ($umol \cdot mol^{-1} \cdot s^{-1}$) =*$\Delta$ CO_2 ppm over 124 seconds ($umol \cdot mol^{-1}$) ÷ the measurement time (124 seconds)* *Equation C16*

For converting this rate of change of the CO_2 ppm value to the soil CO_2 flux ($g \cdot m^{-2} \cdot h^{-1}$), the ideal gas law should be invoked:

PV= nRT *Equation C17*

Thus, the soil CO_2 flux ($g \cdot m^{-2} \cdot h^{-1}$) could be obtained as:

The soil CO_2 flux $(\mathrm{g} \cdot m^{-2} \cdot h^{-1}) = \frac{P \times V}{R \times T} \times \frac{the\ rate\ of\ soil\ CO_2\ change\ (from\ Equation\ C16)}{A} \times 44 \times 3.6 \times 10^{-3}$ *Equation C18*

P: atmospheric pressure (Pa). In this study a value of 100510.78 Pa was used.
V: the volume of the static chamber (0.001257 m^3) (Figure C. 3)
R: ideal gas constant 8.314 $m^3 \cdot Pa \cdot K^{-1} \cdot mol^{-1}$
T: soil temperature (Kelvin) which can be determined from the soil temperature probe inserted to the 10 cm soil depth. (Figure C. 3)
A: the surface area of the static chamber (0.0079 m^2)
44: the molecular mass of carbon dioxide ($g \cdot mol^{-1}$)
3.6: Converts units from $ug \cdot s^{-1}$ to $mg \cdot h^{-1}$
10^{-3}: Converts units from $mg \cdot h^{-1}$ to $g \cdot h^{-1}$.

The CO_2 flux can be translated into a CO_2-C flux by multiplying with 12 and dividing by 44, to account for the differences in the C and CO_2 molar weights.

Production Parameters	Unit	Mean
Nominal HTT	°C	700
Reactor wall temp.	°C	700
Max. char HTT	°C	668
Heating rate	$°C \cdot min^{-1}$	79
Kiln residence time	min	15
Mean time at HTT	min	6
Biochar yield	wt%	23.54

Table C. 2 Details of the pyrolysis process for producing wheat straw biochar pellets.
HTT: highest treatment temperature. Data from the UK Biochar Research Centre (2014).

Parameter	Unit	Mean	SD
Moisture	wt %	2.17	0.22
C total	wt%	69.04	1.32
H	wt%	1.18	0.04
O	wt%	5.3	1.06
H: C_{total}	Molar ratio	0.2	0.01
O: C_{total}	Molar ratio	0.06	0.01
Total ash	wt%	23.82	2.33
Total N	wt%	1.32	0.03
pH	-	10.03	0.19
Electric conductivity	$dS \cdot m^{-1}$	1.52	0.42
Biochar C stability	% C-basis	100.97	0.21

Table C. 3 Basic characterization of the wheat straw biochar.
SD: Standard Deviation. Data from the UK Biochar Research Centre (2014).

	Soil Carbon Category (One-way ANOVA)					
BC vs WP	Total carbon storage of soil	Organic carbon storage of soil	Inorganic carbon storage of soil	$\delta^{13}C_{org}$	$\delta^{13}C_{carb}$	$\delta^{18}O_{carb}$
0-15 cm Soil depth	<0.001	0.025	0.251	0.092	0.804	0.007
15-30 cm Soil depth	0.933	0.689	0.751	0.025	0.706	0.630

BC vs WP	Soil water content (Fisher's Least Significant Difference (LSD) test)
10 cm Soil depth	<0.001
50 cm Soil depth	<0.002
80 cm Soil depth	<0.001

BC vs WP	Soil temperature (Fisher's Least Significant Difference (LSD) test)
10 cm Soil depth	0.887
50 cm Soil depth	0.004
80 cm Soil depth	0.357

BC vs WP	Electricity Conductivity (Fisher's Least Significant Difference (LSD) test)
10 cm Soil depth	<0.001
50 cm Soil depth	0.001
80 cm Soil depth	0.137

BC vs WP	**CO_2-C flux (One-way ANOVA)**			
10 cm Soil depth	0.219			
	CO_2-C Concentration (Fisher's Least Significant Difference (LSD) test)			
10 cm Soil depth	0.797			
35 cm Soil depth	0.988			
50 cm Soil depth	0.478			
75 cm Soil depth	0.5			
	Leachate (One-way ANOVA)			
	Dissolved carbon	Dissolved organic carbon	pH	Alkalinity
BC vs WP	0.60	<0.001	<0.001	0.641
	Grass (t-test)			
BC vs WP	0.488			

Table C. 4 Statistical analysis (*p* value, each test is marked in the table) of data for the soil carbon, water content, CO_2-C flux from various soil depths from the two lysimeters.
BC: lysimeter with wheat straw biochar pellets; WP: lysimeter with wheat straw biomass pellets. Statistical significance is acknowledged when $p<0.05$.

1. **CO_2 concentration (Tukey's HSD)**

		10 cm Soil	35 cm Soil	50 cm Soil	75 cm Soil	Within the whole of group
BC	10 cm Soil	\	0.018	<0.001	<0.001	<0.001
	35 cm Soil	0.018	\	0.12	0.057	
	50 cm Soil	<0.001	0.12	\	0.716	
	75 cm Soil	<0.001	0.057	0.716	\	
WP	10 cm Soil	\	0.707	0.082	0.116	0.212
	35 cm Soil	0.707	\	0.169	0.229	
	50 cm Soil	0.082	0.169	\	0.86	
	75 cm Soil	0.116	0.229	0.86	\	

2. **The relationship between leaching carbon and environmental temperature (Tukey's HSD)**

	Dissolved total carbon	Dissolved organic carbon
BC	0.22	0.62
WP	<0.001	0.026

3. **Paired samples correlations between CO_2 gas and water content**

	10 cm Soil (CO_2-C flux)	35 cm Soil (CO_2-C ppm)	50 cm Soil (CO_2-C ppm)	75 cm Soil (CO_2-C ppm)

BC	Pearson correlation	-0.019	0.05	-0.138	-0.515
	p-value	0.01	<0.001	<0.001	<0.001
WP	Pearson correlation	-0.542	0.174	0.015	-0.109
	p-value	<0.001	<0.001	<0.001	<0.001

1. **Paired samples correlations between CO_2-C concentration and soil temperature**

		10 cm Soil	35 cm Soil	50 cm Soil	75 cm Soil
BC	Pearson correlation	0.267	0.608	0.009	0.448
	p-value	<0.001	<0.001	<0.001	<0.001
WP	Pearson correlation	0.6	-0.03	0.147	0.268
	p-value	<0.001	<0.001	<0.001	<0.001

2. **The relationship between CO_2-C flux and soil temperature at 10 cm soil (t-test)**

BC		<0.001
WP		<0.001

3. **The relationship between oxygen isotope and carbon isotope from the carbonate in soils (regression analysis)**

BC&WP	Pearson correlation	0.580
	p-value	<0.001

Table C. 5 Statistical significance (*p* value, each test is marked in the table) between different depths about soil carbon, water content, soil temperature and CO_2-C flux in the single lysimeter.
BC: lysimeter with wheat straw biochar; WP: lysimeter with wheat straw pellets. Statistical significance is acknowledged when $p<0.05$.

Lysimeter with Wheat Straw Biochar				Lysimeter with Wheat straw pellets			
Collection Date	Item	Result $\delta^{13}C_{org}$	Fraction	Collection Date	Item	Result $\delta^{13}C_{org}$	Fraction
	Wheat Straw Biochar	**-29.53**			**Wheat Straw Pellets**	**-28.95**	
19/06/2018	30 cm soil	-23.94		19/06/2018	30 cm soil	-23.94	
19/06/2018	50 cm soil	-24.55		19/06/2018	50 cm soil	-24.55	
16/08/2018	top soil_BC	-25.44	0.27	16/08/2018	top soil_WP	-24.68	0.15
28/09/2018	top soil_BC	-25.76	0.33	28/09/2018	top soil WP	-24.86	0.19
21/02/2019	top soil_BC	-25.91	0.35	21/02/2019	top soil_WP	-25.91	0.39
04/06/2019	top soil_BC	-26.56	0.47	04/06/2019	top soil_WP	-24.97	0.21
19/08/2019	top soil_BC	-25.78	0.33	19/08/2019	top soil_WP	-25.16	0.24
10/01/2020	top soil_BC	-25.16	0.22	10/01/2020	top soil_WP	-24.88	0.19
15/08/2020	top soil_BC	-23.98	0.01	15/08/2020	top soil_WP	-24.53	0.12
25/02/2021	top soil_BC	-24.98	0.19	25/02/2021	top soil_WP	-24.66	0.15
25/06/2021	top soil_BC	-24.81	0.16	25/06/2021	top soil_WP	-24.46	0.10
Total Average			**0.26**	**Total Average**			**0.19**
16/08/2018	50 cm soil_BC	-25.52	0.19	16/08/2018	50 cm soil_WP	-24.53	0.00
04/06/2019	30 cm soil_BC	-25.38	0.17	04/06/2019	30 cm soil_WP	-24.03	-0.12
19/08/2019	30 cm soil_BC	-25.01	0.09	19/08/2019	30 cm soil_WP	-24.53	-0.01
10/01/2020	30 cm soil_BC	-24.56	0.00	10/01/2020	30 cm soil_WP	-24.48	-0.02
15/08/2020	30 cm soil_BC	-24.53	0.00	15/08/2020	30 cm soil_WP	-24.25	-0.07
25/02/2021	30 cm soil_BC	-25.05	0.10	25/02/2021	30 cm soil_WP	-24.08	-0.11
25/06/2021	30 cm soil_BC	-24.13	-0.08	25/06/2021	30 cm soil_WP	-23.93	-0.14
Total Average			**0.07**	**Total Average**			**-0.07**

Table C. 6 The average fractions of soil organic carbon that was derived from additives in two lysimeters.
BC: lysimeter with wheat straw biochar; WP: lysimeter with wheat straw pellets.

Manually Installed tubes

Date		26/08/2018	28/09/2018	27/11/2018	28/01/2019	21/02/2019	25/03/2019	25/04/2019	21/06/2019	24/07/2019	29/08/2019	30/09/2019	30/10/2019	21/11/2019	11/12/2019	20/01/2020	22/02/2020
Lysimeter BC	Mean	0.09	0.01	0.08	0.01	0.09	0.10	0.32	0.04	0.20	0.08	0.01	0.10	0.06	0.05	0.07	0.07
	SD	0.00	0.00	0.01	0.02	0.01	0.01	0.01	0.03	0.04	0.01	0.00	0.00	0.01	0.00	0.00	0.00
Lysimeter WP	Mean	0.25	0.02	0.00	0.01	0.01	0.00	0.03	0.01	0.34	0.01	0.01	0.01	0.01	0.00	0.00	0.00
	SD	0.01	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Portable CO_2 gas analyser

Date		15/03/2019	26/03/2019	25/04/2019	04/06/2019	21/06/2019	24/07/2019	19/08/2019	19/09/2019	19/10/2019	21/11/2019	11/12/2019	20/01/2020	25/02/2020
Lysimeter BC	Mean	0.05	0.06	0.03	0.01	0.03	0.05	-0.01	0.02	-0.02	-0.05	-0.02	-0.02	0.00
	SD	0.05	0.06	0.06	0.02	0.01	0.05	0.09	0.02	0.07	0.05	0.02	0.03	0.03
Lysimeter WP	Mean	0.02	0.05	0.12	0.05	0.07	0.13	0.05	0.09	0.11	0.08	0.04	0.00	-0.02
	SD	0.01	0.02	0.03	0.07	0.04	0.01	0.05	0.05	0.05	0.05	0.02	0.01	0.07

Table C. 7 Monthly CO_2-C flux ($g \cdot m^{-2} \cdot h^{-1}$) measured from the manually installed tubes and EGM-4 gas analyser, respectively. BC: lysimeter with wheat straw biochar; WP: lysimeter with wheat straw pellets. SD: standard deviation.

	Parameter	Result	Compliant with multi purpose range? (Y/N)	Compliant with specific purpose range? (Y/N)				
				Acid.	Calc	Low F.	Low F. acidic	Low F. calc.
Texture								
Clay content %	$	8						
Silt content %	$	15						
Sand content %	$	40						
Soil texture - (see figure 1)	$	Loamy Sand						
Soil organic matter content % (varying with clay content)								
Clay 5-20%	DETSC 2002#	3.6	Y	Y	Y	Y	Y	Y
Clay 20-35%	DETSC 2002#		N	N	N	N	N	N
Maximum coarse fragment - Content % m/m								
>2 mm	$	37	N	N	N	N	N	N
>20 mm	$	0	Y	Y	Y	Y	Y	Y
>50 mm	$	0	Y	Y	Y	Y	Y	Y
Soil pH value	DETSC 2008#	8.4	Y	N	Y	Y	N	Y
Carbonate (Calcareous only) %	DETSC 2005	0	N/A	N/A	N	N/A	N/A	N
Available plant nutrient content								
Total Nitrogen %	DETSC 2121*	0.17	Y	Y	Y	N/A	N/A	N/A
Extractable phosphorous mg/l	DETSC 2301*	15	N	N	N	Y	Y	Y

Extractable potassium mg/l	DETSC 2301*	20.3	N	N	N	N/A	N/A	N/A
Extractable magnesium mg/l	DETSC 2301*	27.8	N	N	N	N/A	N/A	N/A
Carbon: Nitrogen ratio		12.28	Y	Y	Y	Y	Y	Y
Electrical Conductivity	DETSC 2009	3700	Y	N/A	N/A	N/A	N/A	N/A
Phytotoxic contaminants (by soil pH) mg/kgDS								
Zinc (Nitric acid extract)	DETSC 2301*	139.1	Y	Y	Y	Y	Y	Y
Copper (Nitric acid extract)	DETSC 2301	24.6	Y	Y	Y	Y	Y	Y
Nickel (Nitric acid extract)	DETSC 2301	46	Y	Y	Y	Y	Y	Y
Visible contaminants % m/m								
>2 mm	*	0	Y	Y	Y	Y	Y	Y
of which plastics	*	0	Y	Y	Y	Y	Y	Y
..man made sharps	*	0	Y	Y	Y	Y	Y	Y

Table C. 8 The texture of raw soil in lysimeters. Analysis was processed by Derwentside Environmental Testing Services Limited (Durham, United Kingdom).

$: completed by approved subcontractor

*: unaccredited test

		Specific purpose topsoil			
Parameter	Multipurpose Topsoil	Calcareous	Low fertility	Low fertility acidic	Low fertility calcareous
Soil texture <2mm fraction % m/m					
Clay content %		10 to 35			
Silt content %		0 to 65			
Sand content %		35 to 85			
Maximum course fraction % m/m					
>2 mm		30			
>20mm		10			
>50mm		0			
Mass loss on Ignition %					
Clay 5% to 20%	3 to 20	3 to 20	2 to 20	2 to 30	2 to 20
Clay 20% to 35%	5 to 20	5 to 20	2 to 20	2 to 30	2 to 20
Soil pH	5.5 to 8.5	7.5 to 9.0	3.5 to 9.0	3.5 to 5.5	7.5 to 9.0
Carbonate % m/m		>1			>1
Plant nutrient content					
Total nitrogen % m/m	>0.15	>0.15	-	-	-
Extractable phosphate mg/l	16 to 140	16 to 140	≤20	≤20	≤20
Extractable potassium mg/l	121 to 1500	121 to 1500	-	-	-
Extractable magnesium mg/l	51 to 600	51 to 600	-	-	-
Carbon : Nitrogen ratio	<20:1	<20:1	<35:1	<35:1	<20:1
Electrical conductivity μS.cm-1		If greater than 3 300, carry out exchangeable sodium			
Multi purpose and specific purpose topsoils					
Potentially Phytotoxic elements					

(mg/kg dry basis)	Soil pH <6.0	Soil pH >7.0			
Zn	<200	<300			
Cu	<100	<200			
Ni	<60	<110			
Visible contaminants %m/m					
of which plastics	<0.5				
Sharps, number	<0.25				

Table C. 9 Threshold Values of Raw Topsoil in lysimeters. Analysis was processed by Derwentside Environmental Testing Services Limited (Durham, United Kingdom)

Test	Method	LOD	Units	
Metals				
Arsenic	DETSC 2301#	0.2	mg/kg	8.1
Cadmium	DETSC 2301#	0.1	mg/kg	0.2
Chromium	DETSC 2301#	0.15	mg/kg	12
Copper	DETSC 2301#	0.2	mg/kg	28
Lead	DETSC 2301#	0.3	mg/kg	180
Mercury	DETSC 2325#	0.05	mg/kg	0.08
Nickel	DETSC 2301#	1	mg/kg	13
Selenium	DETSC 2301#	0.5	mg/kg	< 0.5
Zinc	DETSC 2301#	1	mg/kg	71
Inorganics				
pH	DETSC 2008#			7.7
Sulphate Aqueous Extract as SO4	DETSC 2076#	10	mg/l	66
Petroleum Hydrocarbons				
Aliphatic C5-C6	DETSC 3321*	0.01	mg/kg	< 0.01
Aliphatic C6-C8	DETSC 3321*	0.01	mg/kg	< 0.01
Aliphatic C8-C10	DETSC 3321*	0.01	mg/kg	< 0.01
Aliphatic C10-C12	DETSC 3072#	1.5	mg/kg	< 1.5
Aliphatic C12-C16	DETSC 3072#	1.2	mg/kg	< 1.2
Aliphatic C16-C21	DETSC 3072#	1.5	mg/kg	< 1.5
Aliphatic C21-C35	DETSC 3072#	3.4	mg/kg	39
Aliphatic C5-C35	DETSC 3072*	10	mg/kg	39
Aromatic C5-C7	DETSC 3321*	0.01	mg/kg	< 0.01
Aromatic C7-C8	DETSC 3321*	0.01	mg/kg	< 0.01
Aromatic C8-C10	DETSC 3321*	0.01	mg/kg	< 0.01
Aromatic C10-C12	DETSC 3072#	0.9	mg/kg	< 0.9
Aromatic C12-C16	DETSC 3072#	0.5	mg/kg	< 0.5
Aromatic C16-C21	DETSC 3072#	0.6	mg/kg	2.6
Aromatic C21-C35	DETSC 3072#	1.4	mg/kg	100
Aromatic C5-C35	DETSC 3072*	10	mg/kg	110
TPH Ali/Aro Total	DETSC 3072*	10	mg/kg	140

PAHs				
Naphthalene	DETSC 3303#	0.03	mg/kg	< 0.03
Acenaphthylene	DETSC 3303#	0.03	mg/kg	< 0.03
Acenaphthene	DETSC 3303#	0.03	mg/kg	< 0.03
Fluorene	DETSC 3303	0.03	mg/kg	< 0.03
Phenanthrene	DETSC 3303#	0.03	mg/kg	0.07
Anthracene	DETSC 3303	0.03	mg/kg	< 0.03
Fluoranthene	DETSC 3303#	0.03	mg/kg	0.17
Pyrene	DETSC 3303#	0.03	mg/kg	0.15
Benzo(a)anthracene	DETSC 3303#	0.03	mg/kg	0.09
Chrysene	DETSC 3303	0.03	mg/kg	0.08
Benzo(b)fluoranthene	DETSC 3303#	0.03	mg/kg	0.14
Benzo(k)fluoranthene	DETSC 3303#	0.03	mg/kg	0.05
Benzo(a)pyrene	DETSC 3303#	0.03	mg/kg	0.09
Indeno(1,2,3-c,d)pyrene	DETSC 3303#	0.03	mg/kg	0.05
Dibenzo(a,h)anthracene	DETSC 3303#	0.03	mg/kg	< 0.03
Benzo(g,h,i)perylene	DETSC 3303#	0.03	mg/kg	0.06
PAH - USEPA 16, Total	DETSC 3303	0.1	mg/kg	0.95
Phenols				
Phenol - Monohydric	DETSC 2130#	0.3	mg/kg	1.0
Asbestos Analysis				
No Asbestos detected				

Table C. 10 Chemical analysis of soil samples from lysimeters. Analysis was processed by Derwentside Environmental Testing Services Limited (Durham, United Kingdom)

#: MCERTS (accreditation only applies if report carries the MCERTS logo).

*: not accredited

Figure C. 1 Image of a double ring infiltrometer used in the lysimeters. (Image Jiaqian Wang)

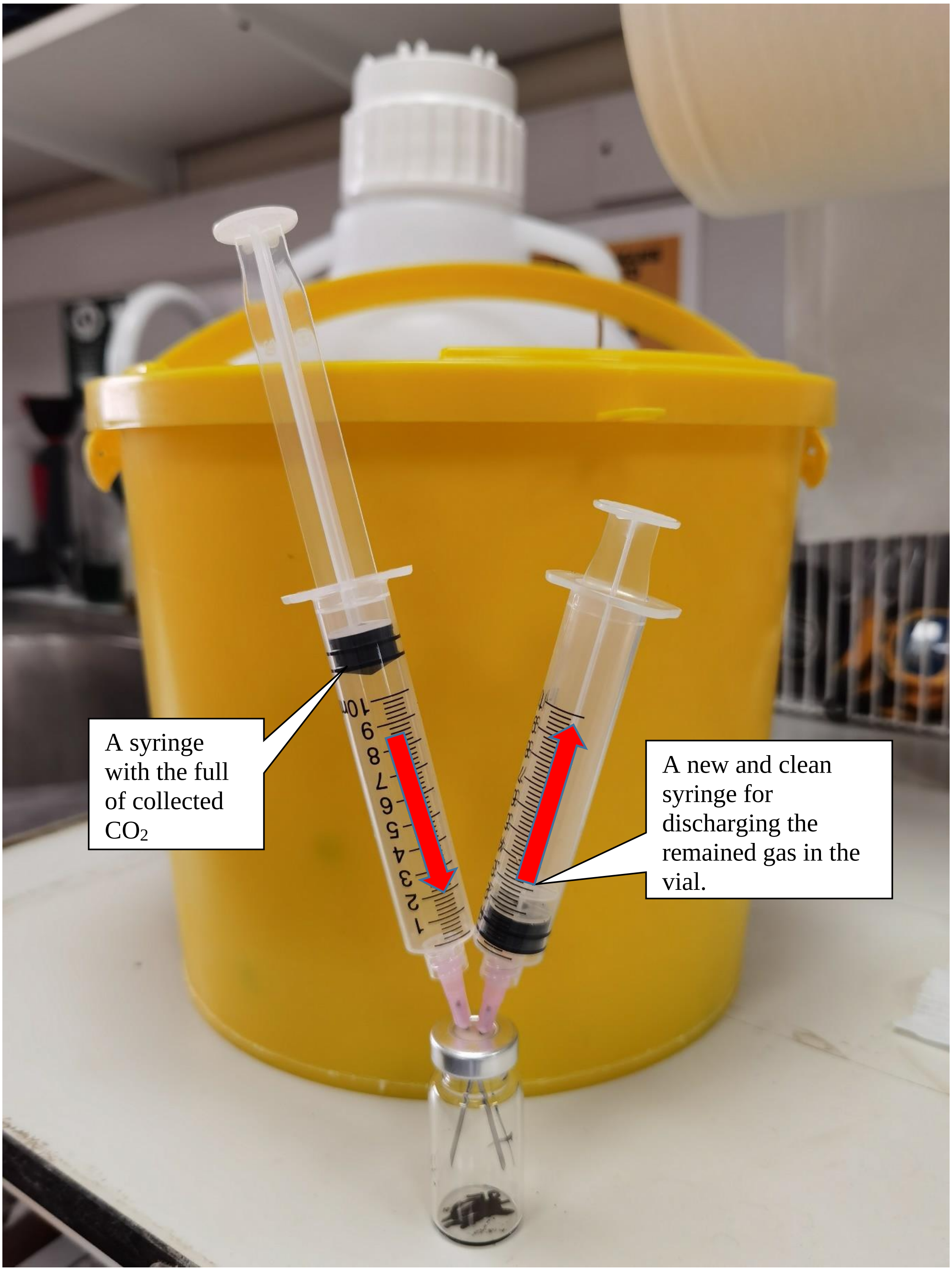
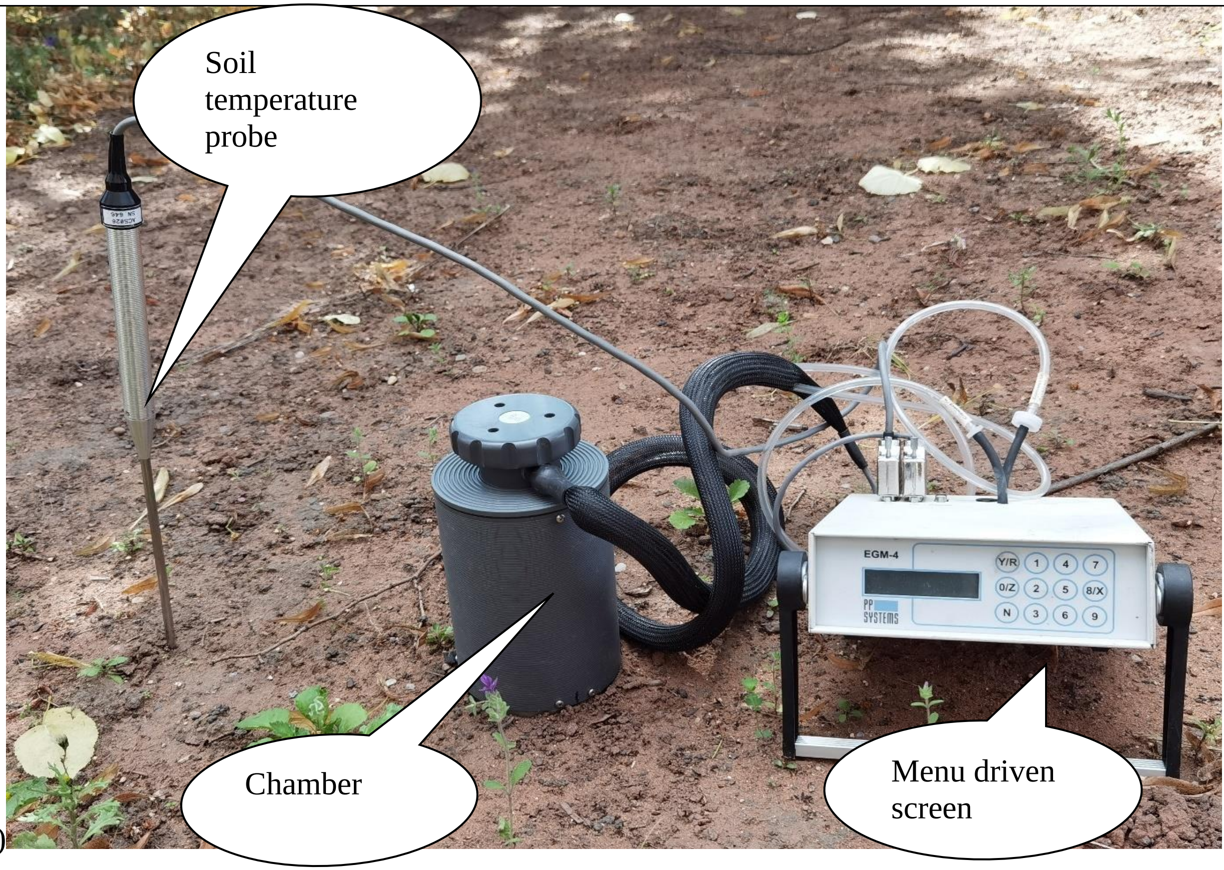
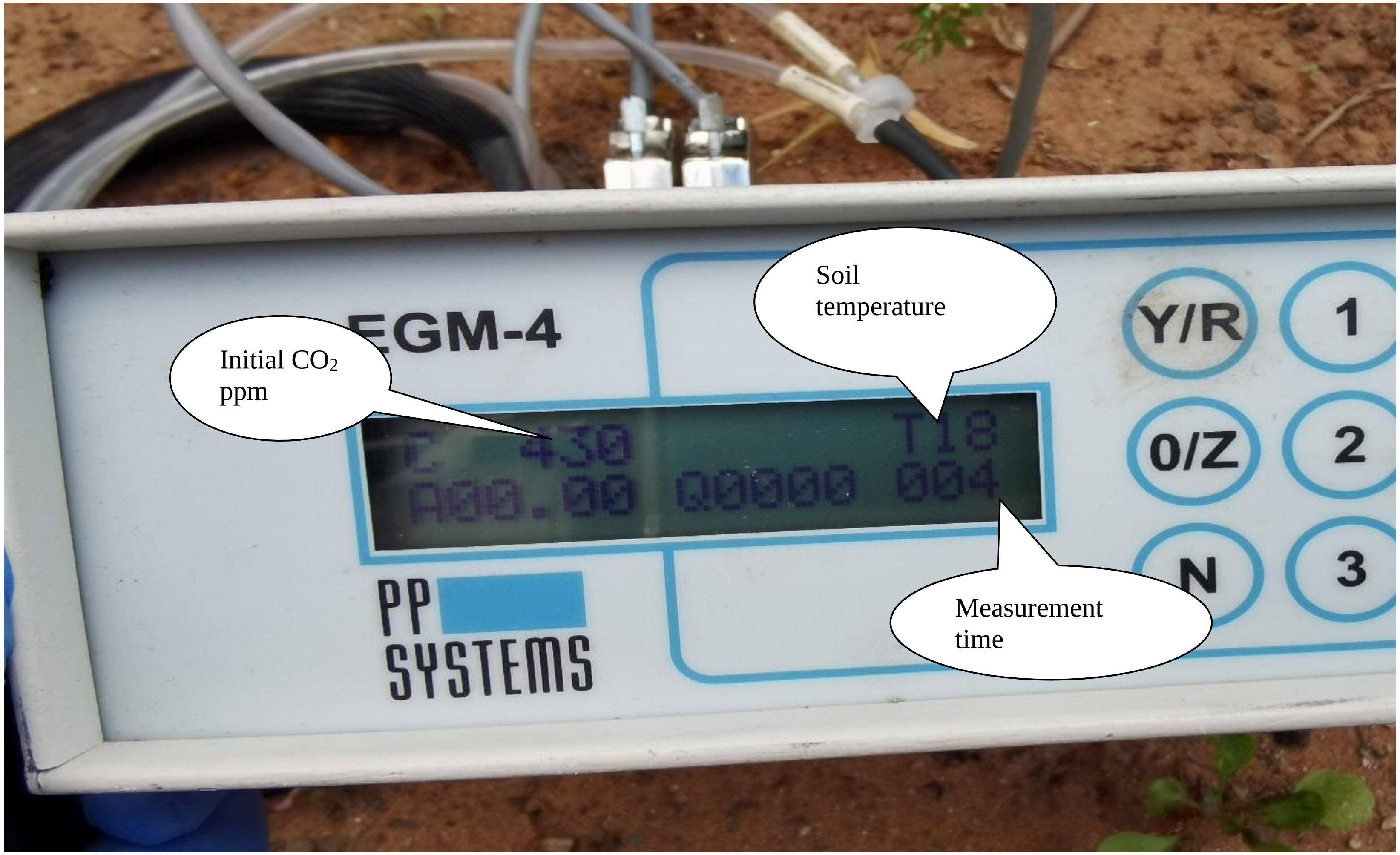


Figure C. 2 The device of collecting CO_2 emissions manually from the lysimeter soils. (Image Jiaqian Wang)


Soil temperature probe
Chamber
Menu driven screen
EGM-4
PP SYSTEMS

a)

b)


Initial CO_2 ppm
Soil temperature
Measurement time
EGM-4
C 430 T18
A00.00 Q0000 004
Y/R
1
0/Z
2
N
3
PP SYSTEMS

c)

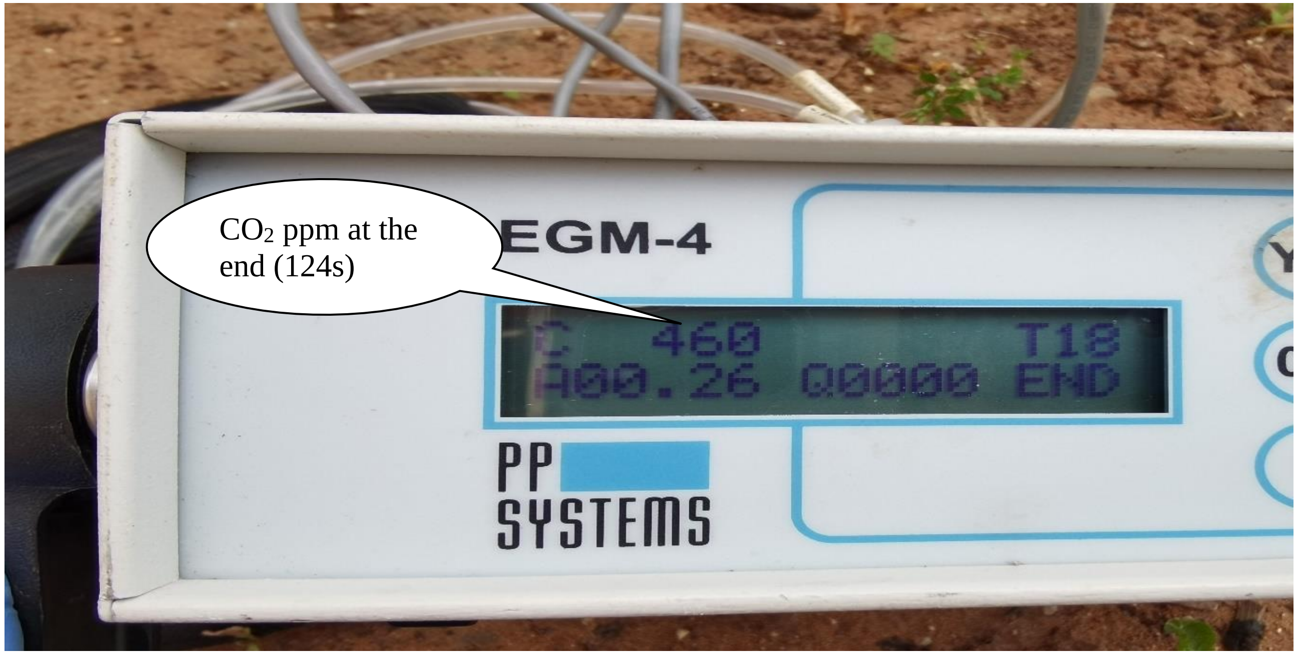


Figure C. 3 How to set up the EGM-4 CO_2 Analyzer and how to read the value of CO_2 ppm. (Image Jiaqian Wang)

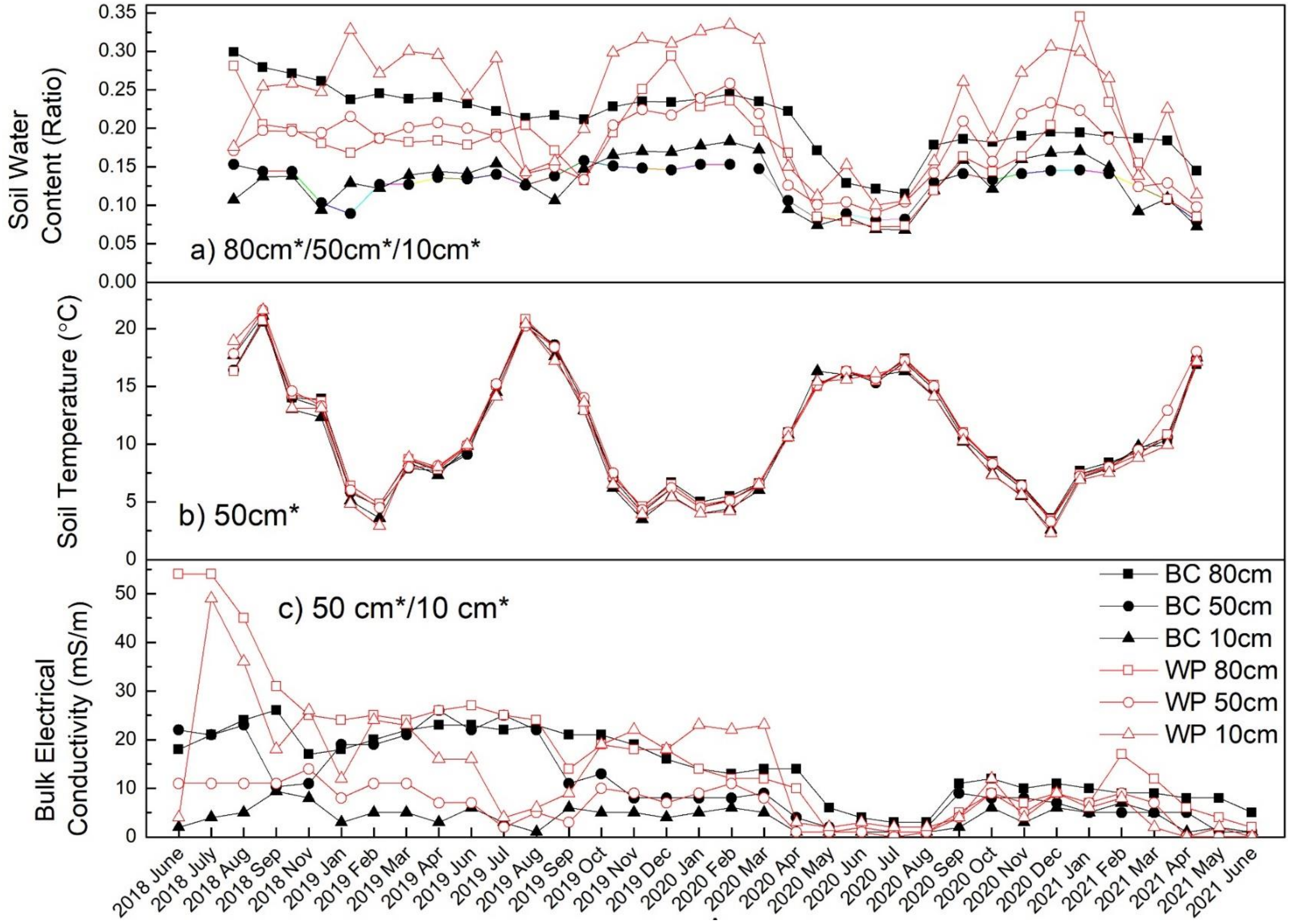


Figure C. 4 Monthly mean soil water content, soil temperate and soil bulk electrical conductivity at 10 cm, 50 cm, 80 cm depth in response to soil additives.

BC: lysimeter with wheat straw biochar; WP: lysimeter with wheat straw pellets. *: the statistical relationship exists between two treatments at specific soil depth.

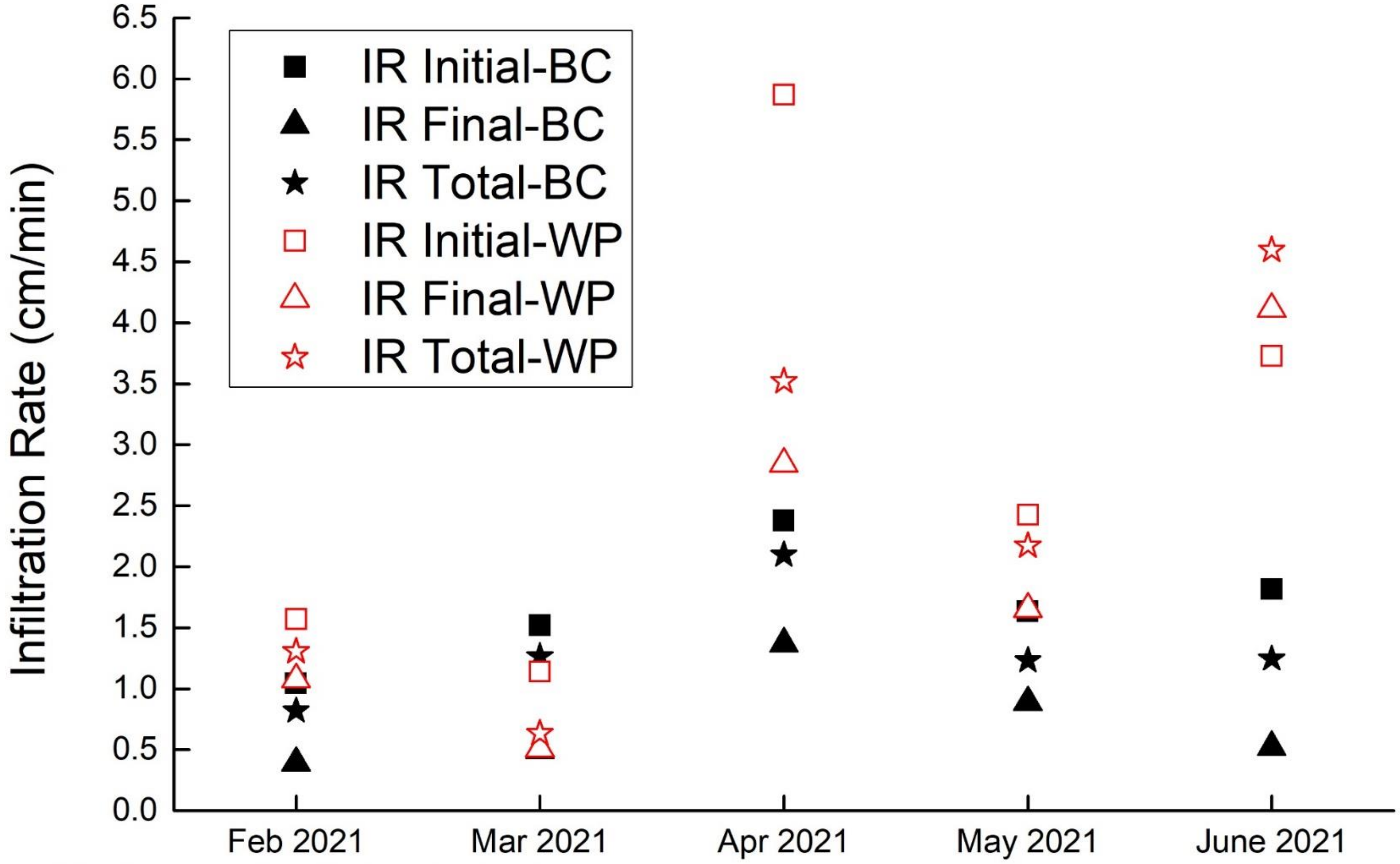


Figure C. 5 The infiltration test results from February to June 2021.
BC: lysimeter with wheat straw biochar; WP: lysimeter with wheat straw pellets. IR Initial: the infiltration rate in the first minute. IR Final: the infiltration rate in the last minute. IR Total: the average infiltration rate during the whole process.

Appendix D

Appendix D1. Notes from the interview with Farms Director in Newcastle University

1. **Who decides what crops are grown at Nafferton Farm and Cockle Park Farm? How is the crop rotation decided?**

Reply: In principle, the Farms Director decides crop species and certainly the Director will discuss with managers of the arable, pig and dairy operations to confirm the crop species selected would satisfy the requirements they ask. In addition, the Director will consider any experiment trials are needed, which do not always provide plenty of time for managers to consider properly.

A couple of factors would influence the decision of choosing crops: making money, meeting the straw and feed requirements for the livestock on the farms, and the latest farm strategy aiming at “zero-till” which includes the assessment of environment and carbon. Especially due to the update of the strategy, farm managers change and track the crop rotation as required.

2. **How are you familiar with “Carbon Management Plan” of Newcastle University?**

Reply: The Director has not read the plan in detail but has attended some seminars mentioning carbon management of farms. The Director has regular conversations with a leader from Sustainability Team and both of them agree that farms should be included in the plan.

At present, two farms have not been involved in Carbon Management Plan of Newcastle University, although they cover 2,000 acres (800 ha), and thus the farms have to develop their own internal “Carbon Strategy”. This strategy is in development, which based on a previous 3-years project focusing on the finances, the development goal of the farms, contracts, properties, etc, but nowadays this project could think more in terms of carbon sequestration. Due to the COVID-19, drafting the carbon strategy is being a little delayed but the Farms Director plans to construct all the ideas during the next 6 months, and welcomes comments from people who are related with or interested in it. Furthermore, the carbon calculator is applied to measure how much carbon is sequestered in the farms, as part of a Student Office of Sustainability project which addresses this, and how to pay farmers for carbon sequestration.

In this "Carbon Strategy", putting the farms back onto the right footing is a priority task, which needs many metrics, and this takes a lot of time and effort to accomplish including setting up the work system. Then, carbon sequestration in the environment and animal management would be considered in the strategy of farms.

3. **What is the responsibility of farm managers in actioning the "Carbon Management Plan"? Are there any carbon capture projects at the university farms?**

Reply: The farm management team uses less diesel, and has restructured the business in other ways that lead to carbon savings. It has started carbon calculation to check the metrics relating to improved carbon sequestration as a consequence of changes they have made. A carbon calculator named "Farm Carbon Toolkit" and "Agrecalc", will be used due to its characteristics of simplicity and free use across the whole farm's complex operations. "Cool Farm" is not appropriate for a multi-enterprise farm such as ours.

In terms of carbon capture projects, a small scale one was conducted in Cockle Park Farm. Once it is successful, this scheme will be extended at Cockle Park Farm. Other carbon reduction activities are being carried out as well, such as cutting soya from the dairy ration for the last 12 months, talking with nutrition companies about taking soya from the pig diet, looking the way to improve the protein content of grain in the farms, and altering the cultivation system to reduce carbon loss from soils. Moreover, managers assess the costs in terms of farm spend and the productivity of the soil.

4. **My research found that topsoil carbon storage was 41% significantly higher in permanent grassland as compared to arable land. In addition, I looked at the Cockle Park Farm land use history, and found the percentage of land managed as pasture decreased from 84% in 1900 to 21% at present, which according to my estimates caused a total carbon loss of 3,251 tonnes. In your opinion, would it be desirable and feasible to convert more land at Cockle Park Farm from arable lands back into pasture for improving carbon storage?**

Reply: The farm manager might accept the suggestion in terms of carbon storage but would not think about it from the financial perspective, unless there is a good economic or research reason and the change could be funded. The management of permanent grassland in Cockle Park Farm to increase capacity is controlled by investment in livestock facilities, slurry handling, infrastructure availability, the types of enterprises introduced, etc. Converting more land into pastures would decrease the efficiency of the farms accompanying with the loss of productivity which has to gain elsewhere.

In the future, more organic matter, from catch crops, cover crops and companion crops, might be put in the fields across Cockle Park Farm. Furthermore, Cockle Park Farm would not be a sole working place to consider its grassland conversion proportion, because managers would look at the whole farm fields instead of the specific one.

5. **From my study, the most effective way of increasing carbon on the university estate is by converting crop fields into woodland. The carbon content of woodland soil is around 4 Kg/m^2 greater than in crop fields, and with the additional carbon in tree biomass, woodlands could store 15-16 Kg/m^2 more carbon than crop fields. In your opinion, would it be desirable and feasible to convert more land at the farms from crop fields back into woodlands for improving carbon storage?**

Reply: There are several challenges, and one of them is the tenancy contract: there are 500 hectares at Nafferton Farm & Ousten Farm that cannot be converted to woodlands, because of the terms of the tenancy. At Cockle Park Farm, the terms of the lease by which the University holds the farm may not permit increasing woodland area. If our organisation were a business or company and owned the land, we would be free to choose, but it will change tax status, inheritance tax status, and other significant financial/legal factors. Consequently, the conversion from crop fields to woodlands is not viable unless a substantial subsidy could be received. In terms of carbon, there was a debate in the farm meetings concerning the value of woodland: the capability of carbon storage of trees varies at their different life stages, and carbon capture might not be as valuable as first thought.

The Director has previous experience with managing a big area of woodlands and acknowledges that this is not easy and not a good way to make money.

6. **We know that many farms are adopting agroforestry as a production system. Do you have plans to do this? If so, how do they relate to the management of soil carbon? At each farm, how many hectares might be converted to agroforestry?**

Reply: Agroforestry practice is being planned in one site at Cockle Park Farm which covers around 22 hectares. This is a complicated work because farm managers have to figure out the plan of the practice, what sort of species will be used, etc. Some similar works have been conducted and perform well.

7. **To what extent do you think the farm management team would be willing to produce biochar from crop residues at the university farms to augment carbon in soil on the farm and around the campus? Which farm residue would be most suitable for the production of biochar? Would you have any concerns?**

Reply: The Farms Director has not considered biochar yet. No crop residues would be fit for the purpose of biochar production in our farms because animals need crop residues as well and not sufficient residues remained to be made into biochar. Regarding of the cultivation system of the farms moving towards, farm technicians try to not disturb the crop surfaces. If they do not need wheat straw, farm technicians would use a strip header to take the head of wheat straw off and leave the main part of straw standing in the fields, so straw is able to break down naturally to the soils which benefits draining and feeds organisms.

Also, the Farms Director is willing to build links with other organisations in biochar research. Another thing that the farm managers are working is to build up the business connections of farms in a more strategic way, so to make it easier for staff and students to find the right partner for cooperation.

8. **There are some areas in Cockle Park Farm are marked as grassland which seldom are ploughed in the past 5 years at least, how often do you mow the pasture and how do you dispose grass residues in these areas?**

Reply: Grass is managed within the rotation grazing system. In Nafferton Farm, farm managers measure the dry matter of grass weekly which set the grazing platform management. In Cockle Park Farm, the grazing practice could be called as “set-stock”: a group of animals of set number live in the pastures in spring, so the grass is consumed and sometimes farm technicians need to supplement it. Plus, in Cockle Park, more infrastructure will be put to pastures to move to rotational grazing to avoid grass waste.

As for animal residues, farm managers record and track the slurry being applied and measure NPK, sulphate and other nutrients. This approach helps managers deciding what the exact amount of nutrients farms need rather than estimation. Hopefully, nutrients could be saved efficiently in the future.

Appendix D2. Notes from the interview with ground manager in Newcastle University

1. **When do you plan tree planting around campus, what types of tree species do you choose and why?** **Reply:** Many factors affect the choice of tree species. Firstly, it depends why you put trees in the place; whether it is for street replacement; whether the previous tree was diseased; whether the tree was a memorial tree; the size of the area to put a tree in (large space or small); shape and crown of trees; the possibility to survive in the university's environment. An example: if planting a memorial tree, estate team would be interested in some species which could show interest throughout a year as long as possible, such as cherry trees or other species with the similar nature. Therefore, there are several interests for the whole seasons: the blossom in spring, fruits in summer and awesome leaf colour in autumn. In addition, if there is a tree in disease, estate team may not put the same tree back for avoiding the same thing happens again and will figure out the factor resulting in the tree death, and then find different tree species. Moreover, Newcastle University locates in the city centre, so there are not many spaces to put trees and the estate just try to fit everything they can. For instance, it is less likely to plant huge English Oak which will occupy much area. Estate team has to think what trees will look like and how its shape develops in 20-50 years combined with the distance to the buildings nearby the trees. Furthermore, survival is significant, because some tree species might perform pretty well in other places but find it hard to survive in a particularly demanding area like north-eastern England.
2. **What are the specific tree species most widely favoured by estate managers?** **Reply:** Part of the answer is included in the reply for question 1. Also, estate managers would like to choose which tree species is fashionable to plant, any popular tree types regarding of designing landscape? During the last 12-13 years, a lot of fruit trees including cherry trees, pear trees and apple trees were planted in Newcastle University. Particularly, lots of pear trees were introduced recently: a popular one is

"Pyrus calleryana Chanticleer" which is quite close to being an evergreen tree and whose leaves appear not to fall in cold seasons, so this tree species is a good option in Newcastle University to enrich the natural colour because there are not so many evergreen trees on campus in winter.

3. **After removing dead trees, how do you choose a new replacement? Is that the same as the previous tree species? And in the same location? How often do you assess the health and risks of trees?**

Reply: In the same location, the estate team prefers to put the similar tree species back but it still depends on buildings around the place and ground conditions. For instance, some cherry trees which grown previously leave a hard, long and complex root net. This increases the difficulty for ground managers to dig a reasonable tree pit to plant a successor. The managers always try to choose a tree which is most close with the one growing at the original place but sometimes it is inevitable to make a shift.

As for assessing the risks and health of trees, in practical terms, the team managers always check the health of trees when they walk around the campus. For instance, in summer and spring, it is easy to spot something wrong in the trees if leaves do not grow in some certain areas or there is an unusual change of leaf colour. Additionally, the trees on campus are informally inspected every 18 months, to alternate when they are in full of leaves (summer) and bare (winter), and thus managers are able to discover the difference of tree conditions between two inspected dates and diagnose tree disease and disorder growth.

4. **If there is a particular tree species which could store more carbon per m^2 than other tree species, are you willing to preferentially choose it?**

Reply: If there are practical data to prove one species could store more carbon, the grounds manager would like to try it, absolutely, but firstly will balance any sort of performance of trees. It would not happen to plant trees just because of their carbon storage potential, especially planting the same tree species at the same area, because there is much evidence about the fast spread of tree disease among the particular tree species around the world and so a single species might be vulnerable. Mixing different tree species, instead, would be an ideal and reasonable plan across the whole site in order to maintain the biodiversity and prevent disease spread within the same plant genus.

5. **Are you familiar with the "Carbon Management Plan" of Newcastle University? If yes, what is the responsibility of the estate team in actioning the "Carbon Management Plan"?**

Reply: The ground manager is familiar with it.

In terms of responsibility on carbon management, the Estates Department uses more petrol and diesel than other parts of the University, and in order to reduce this by using electric vehicles and battery-powered machines. In addition, the estate team could decide which grass seeds or what types of grass used in the sports ground to sequester more carbon. At the meantime, the manager would check how valuable for choosing the alternative grass seeds. The manager is concerned about the carbon footprint, however, grass seed is selected on its performance, turf durability and how it withstands the weather as a priority. In the past several years, carbon is not one of the things to consider from estate's respective, but they would like to think more in the future (not anything too significant yet).

Some comments about the carbon capture garden where it mixes crush rocks and compost: the ground manager would accept this kind of idea to build a carbon capture garden but the key point is to try it in a small scale firstly and make sure there is no harm. If it works well, it is possible to apply across the campus.

How the estate team makes the soils in a football pitch or a pitch from scratch: Combining the soils with sands. There is a fair percentage of soils versus sand to maintain the grassland in an ideal condition, otherwise, too many sands would cause much draining so water and fertilizer would leach straight through; similarly, the ground isn't expected to dry up quickly which is then too hard for a playing surface.

6. **Are you willing to increase the number of trees on campus for augmenting tree biomass and soil carbon? If yes, by how many percent do you think the number of trees on campus should be increased?**

Reply: The manager would like to try it, whereas, balancing the available space is the first step including how to fit the university needs.

A specific number about planting how many trees is difficult to decide. It is less likely to plant a large number of trees for just one goal because our campus is in the city centre. What the estate team can do nowadays is try to fit every available spot one by one, and certainly check the possible shape of trees in the future. Over the campus, in terms of increasing the percentage of trees, the estate team would be struggling with anything higher 5%.

7. **To what extent do you think the estate team would be willing to apply biochar produced at university farms from crop residues to augment carbon in soil around the campus? Would you have any concerns?**

Reply: The manager considers what is that exactly and how the biochar produced. Is it 100% safe? how to store this material? is it stable? Any safety issues from dust? Anything leach out to drainage system? Is the large quantity needed? Does the application need to dig in or just put on the surface?

8. **At Heaton Sports ground, do you leave the grass clippings on the lawn, or if you remove the clippings and compost/dispose somewhere else?**

Reply: Grass residues stay where they are clipped. The managers clip the grass regularly as long as it grows a little higher.

9. **At Heaton, do you know how often ground managers do the grass clipping?**

Reply: The frequency of grass clipping varies throughout the year. In spring and summer, cutting is conducted weekly for 6 months of a year. Either side of the ground is cut every two weeks during three months in autumn. During the other three months in winter, the manager would not clip the grass. Moreover, this kind of schedule

slightly changes yearly due to climate and temperate change and any other factors affecting the grass growth.
10. **At Heaton**, **do managers apply any fertilizer or herbicides to the ground**? **Reply:** The managers apply fertilizer several times throughout a year. There are 2 types of fertilizers: “spring-summer” fertilizer promotes grass and leaf growth; “autumn-winter” fertilizer puts energy to the root develop system. The managers use “broad leaf weed killer” annually and a herbicide named “T-2 GREEN”.
11. Any other ways they used to maintain the good condition of grassland? **Reply:** Horticultural practice is involved to maintain the good condition of the lawn. Some of machines being used with big spikes at the back of the machine, and these spikes are able to punch big holes in the ground, which helps water drainage. Moreover, a large number of particular sands (Mansfield Sand, 90 tonnes at a time and 400-500 tonnes every year approximately) with the right particle size distribution are used throughout a year to improve the drainage. Some sports grounds in the university are located on what was agricultural land. As these have more natural clay contents than appropriate for sports fields the drainage is a primary factor to consider for ground managers.

Appendix D3. Notes from the interview with the Sustainability Team in Newcastle University

1. **The Sustainability Team declared a climate emergency in April 2019; what is the evidence that informed this decision?**

Reply: The most famous and probably the main report in this area is IPCC 2018 where the social change and actions being required to stop global temperature warming: maintaining a degree increase about 1.5°C against pre-industrial level. As for the international agreements, such as the Paris Agreement, a lot of countries including UK have signed. It is written in UK law to achieve net-zero carbon by 2050. As a research intensity institution, Newcastle University wants to make sure that everything the institution owns to put in place, in terms of strategy and decisions, are all evidence-based. In addition, a study done by the Tyndall Centre which was commissioned by Newcastle City Council, looking at how Newcastle as a city would meet the requirements of the Paris Agreement to achieve carbon net-zero by 2050. Furthermore, according to the evidence and recommendations of the report from the study, it shows that the city of Newcastle needs to reduce carbon emissions by 12.8% annually. This is an explicit confirmation that Newcastle University, as part of the city, needs to declare climate emergency because it's imperative that the university acts early to do the right things. Apart from these two main reports, there is a growing amount of science following this area over the past decades. Anecdotally, a lot of extreme weather events happen globally. For example, at the start of 2020, Australian wildfires, severe flooding in the UK, and many other things increase the frequency of extreme weather events. There is substantial evidence to declare the climate emergency from a university's perspective.

2. **In order to accomplish the "Net -Zero" carbon aim by 2030, one of the approaches you have proposed is "land management methods such as tree planting and peatland restoration can be achieved". Are there any related actions being undertaken by the university? In your opinion, are such methods effective in achieving the university's net-zero carbon aim? Does the university have any peatland to restore?**

Reply: The university does not own any peatland. The campus of Newcastle University is located in the city centre, with a couple of satellite sites. The university does not have much forestry, either. In terms of land management, no project has been

undertaken at present with the aim of sequestering carbon, but the Sustainability Team do look a number of avenues both internally and externally, and are interested in this. There is a project led by Newcastle City Council, called "The city community forest", starting to look at tree planting and the university could be involved in this. Additionally, the Sustainability Team looks at a number of credible, local/regional/UK-based offsetting projects as well. Personally, the Energy & Carbon manager would advocate more tree planting which enhances the biodiversity across the campus. For making a real significant impact on the university's carbon emissions, the scale of tree planting in the places is quite substantial. In the 1st stage of the climate action plan of Newcastle University, particularly around net-zero carbon target, driving down carbon emissions rather than extracting carbon from the atmosphere is the primary route. Although tree planting will be considered, the plan will be implemented one year later at least.
3. **Do you think it is desirable to increase the number of trees on campus for augmenting biomass and soil carbon? If yes, by how many percent do you think the number of trees could be increased?** **Reply**. The Energy & Carbon Manager is not familiar with the subject of this question. Certainly, the Manager advocates increasing the number of trees on campus, not just from a carbon reduction perspective, but of all the other advantages that bring in terms of creating an attractive environment and well-being benefits for staff and students. Building a green campus would offer a tangible impact on people's daily life in the university. The available space, particularly on a central campus, that the university owns which could be utilized to impact carbon emission is minimal because its compact nature, and spatial constraints are going to be the key barrier to plant trees. Other areas like the University's farms and sports grounds might could be considered more suitable for tree planting.

4. **We have an interview with farm director about the responsibility of farm on carbon management. The farm in our university is going to frame "Carbon Strategy" at a farm-scale by itself but which hasn't involved in the university's climate action plan, do managers from Sustainability Team consider counting the function of farms into the whole institutional carbon reduction target?**

Reply: "Carbon strategy" of the farms definitely will help the Sustainability Team to manage institutional carbon reduction. There is a balancing act to be struck not just within the farms but across all work of Sustainability Team which has to align with the University's Vision and Strategy, and business needs. The Sustainability Team needs to think other business plans and priorities within the university be the areas of research, different revenue streams, or if they involve other partnerships. From the wide perspective of university, for the conversion from crop to pastures, whether the conversion is the best use of institutional resources, whether staff and student in the university would invest, whether other stakeholders across the university would be happy with this, are three aspects that the manager concerns.

The Sustainability Team have worked closely with the farm directors to look the ways using the farms. Extracting emissions from the atmosphere through planting trees, using the land for renewable energy generation like solar panels, and other methods will certainly be included in the carbon strategy, but all of them are in discussion.

The Sustainability Team includes the farms to calculate the emissions footprint and records energy usage and waste production of the farms. The farm director and the main leader of Sustainability Team will discuss the carbon strategy and what projects will take place at the farms; however, nothing is concrete at present.

5. **My research found that topsoil carbon storage was 41% significantly higher in permanent grassland as compared to arable land. In addition, I looked at the Cockle Park Farm land use history, and found the percentage of land managed as pasture decreased from 84% in 1900 to 21% at present, which according to my estimates caused a**

total carbon loss of 3,251 tonnes. In your opinion, would it be desirable and feasible to convert more land at Cockle Park farm from arable lands back into pasture for improving carbon storage?

Reply: The Energy & Carbon manager would advocate any improvements in carbon management, but similarly, the Sustainability Team needs to consider other stakeholders' opinions, other positive or negative points for the land conversion, balance the changes between land use, and see how that affects the university's business activities.

The Energy & Carbon manager thinks this estimation can form the part of carbon strategy and particularly in terms of reducing the university's Scope 3 carbon emissions. This offsetting methodology could become prevalent across the UK for landowners whose lands are not really in use, so there would be a market for carbon offsetting.

6. **From my study, the most effective way of increasing carbon on the university estate is by converting crop fields on the university-managed farms into pastures or woodland. Including the additional carbon in tree biomass, woodlands could store 15-16 Kg/m^2 more carbon than crop fields. In your opinion, would it be desirable and feasible to convert more land at the farms from crop fields back into woodlands for improving carbon storage?**

Reply: If the amount of carbon per m^2 could be stored more, this approach is obviously beneficial but is necessary to look how the whole of management is going.

7. **Do you think it is a good idea to apply biochar produced at university farms from crop residues to augment carbon in soil around the campus? Would you have any concerns? could you remember some student's programmes which are in within the realm of environmental sustainability? What aspects are these programmes aiming for?**

Reply: it is nice to create circular economy benefits if the university could use something such as food waste or crop residues from the farms to spread across central campus. The manager considers the biochar's impact on flowers, trees, and other vegetations. Additionally, one of the considerations from the manager is the embodied carbon in the biochar. Also, if the biochar cannot be produced at the university, where to purchase biochar, where would biochar be coming to, and how would biochar be transported. Lastly, how often would biochar be recycled generally and when do the managers need to spread it again.

One of the projects that Sustainability Team is starting to take into consideration is capital development which sets an embodied carbon target as part of the climate action plan around how the university refurbishes and builds new buildings.

The Energy & Carbon Manager acknowledges that balancing all areas involved in the carbon management plan is tough: making the right decisions but doing it quickly. Considering the deadline for achieving net-zero carbon is 2030 in Newcastle University, the university would be extremely busy to achieve everything available to meet the requirement. However, reducing carbon cannot be progressed under pressure or rushing into something, which might render more negative outcomes.

Given the previous and current outlook, there are a couple of specific students' programmes studying carbon sequestration across the university and the Sustainability Team assists them every year. The Sustainability Team provides information and data to masters and offers the placement for students to work on environmental and energy management system. Some degree programmes are related to sustainability and carbon reduction, etc.

8. **Due to the national lockdown, the university has gone through a period of low energy consumption, little staff travel and low emissions, and thus are there any new findings in terms of university carbon management?**

Reply: The low energy consumption during the lockdown provides a perfect testbed for identifying where the university's particular electricity baseload comes from. Apparently, the low energy consumed during this period is partly due to 24-hour ventilation provision for labs and fume cupboards, mechanical and ventilated sensitive areas, the running of data centres and computers on campus. Because all of the non-essential stuff was effectively switched off, the Sustainability Team is able to drill down the sources of fundamental baseload. More importantly, the Sustainability Team could know how far the university could meet net-zero carbon, which avoids impacting the central service too much. In addition, as for the heating system, the university can check whether there is a faulty control. There is one building management system across the campus to control heating and ventilation usage, and to make sure buildings are operating correctly. Therefore, the Sustainability Team could identify the areas which are not controlled properly.

As for staff travel, the Team conducts a survey every two years: in one year, the team will do a staff travel survey and the following year will do a student travel survey, and the year after go back to staff travel. However, the response obtained from the survey is quite poor generally. The next survey will be conducted in the first half of 2021, which is the first survey since the lockdown and will bring more fresh data. Since few people travel to work, the carbon produced by staff travel will certainly decrease as the Team expects.