



GROWING LOCAL ENERGY

# The carbon sequestration potential of hedges managed for woodfuel

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## Executive summary

With rising concern over climate change, global efforts are being made to reduce emissions and increase carbon sequestration. Hedgerows – a prevalent feature of the British countryside, with an estimated 700,000 km in Great Britain – provide a multitude of ecosystem services and sequester carbon in both above and below-ground biomass (Falloon et al., 2004; Carey et al., 2008). Despite their potential to store considerable amounts of carbon, little empirical data exists on the role that hedgerows play in capturing and storing carbon (Falloon et al., 2004; Follain et al., 2007). Hedges can also be managed for woodfuel, a renewable source of energy, using practices such as coppicing (Devon Hedge Group, 2014). It is therefore important to ask the question: does coppicing, the cutting and removal of above-ground biomass, have a negative impact on soil carbon stocks and is this balanced by the carbon savings in regrowth and offsetting of fossil fuel use?

To determine the effects of hedgerow management for woodfuel on carbon sequestration, carbon stocks and flows were estimated for paired 15m coppiced and un-coppiced plots established in three hedges of different species in the south of England, and an existing process-based model of the carbon sequestration under short rotation coppice adapted to a woodfuel from hedgerows scenario. The impacts of coppice management on carbon storage were then assessed along with the potential to offset fossil fuel use using a carbon budget analysis.

The study revealed that while hedges which are not managed by coppicing sequester larger quantities of carbon, total carbon savings are higher when hedges are managed by coppicing due to the substitution of fossil fuels via the production of woodfuel. Although the results presented from this small-scale, short-term study should be viewed as provisional, they present a useful starting point for future enquiry, identifying the need for long-term chronological studies and data collection on carbon sequestration processes specific to hedges. Collection of further empirical data on the carbon sequestration potential of hedgerows will be needed to validate existing estimates and models and to inform decisions not only at a farm management level but also for wider policy.

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## 1. Introduction

With rising concern over climate change, global efforts are being made to reduce emissions and increase carbon sequestration. Hedgerows – a prevalent feature of the British countryside, with an estimated 700,000 km in Great Britain – provide a multitude of ecosystem services and sequester carbon in both above- and below-ground biomass (Falloon et al., 2004; Carey et al., 2008). Despite their potential to store considerable amounts of carbon, little empirical data exists on the role that hedgerows play in capturing and storing carbon (Falloon et al., 2004; Follain et al., 2007). Hedges can also be managed for woodfuel, a renewable source of energy, using practices such as coppicing (Devon Hedge Group, 2014). It is therefore important to ask the question: does coppicing, the cutting and removal of above-ground biomass, have a negative impact on soil carbon stocks and is this balanced by the carbon savings in regrowth and offsetting of fossil fuel use?

To determine the effects of hedgerow management for woodfuel on carbon sequestration, carbon stocks were estimated for paired 15m coppiced and un-coppiced plots established in three hedges of different species in the South of England, and an existing process-based model of the carbon sequestration under short rotation coppice adapted to a woodfuel from hedgerows scenario. The impacts of coppice management on carbon storage were then assessed along with the potential to offset fossil fuel use using a simple carbon budget analysis.

The report begins with a literature review of existing information on the carbon sequestration potential of European agricultural hedgerows, critically assessing its successes and limitations. This is followed by a description of the methods used to estimate hedge carbon stores, model the effects of coppice management on carbon storage and to calculate the carbon budget of each scenario. The results and discussion are structured around these three methods: estimated carbon stores, model adaptation, and carbon budgets, and lead to a conclusion based on the report's findings and supporting literature on the carbon sequestration potential of UK hedges managed for woodfuel.

## 2. Literature review

The following review evaluates the current state of research on the carbon sequestration potential of European agricultural hedgerows, critically assessing its successes and limitations. Due to its significant role in the sequestration of carbon and prominence in hedgerow research, studies on soil organic carbon (SOC) provide the main focus of the review.

### 2.1 Carbon stocks and sequestration estimates

Soils contain over three times the quantity of organic carbon found in vegetation and double that of the atmosphere (IPCC, 2000; Alemu, 2014). In forest ecosystems, below-ground carbon accounts for 10 to 46% of the total tree carbon pool (Helmisaari et al., 2002). Increases in net primary production (NPP), both above- and below-ground, increase soil inputs and hence have the potential to increase SOC (IPCC, 2000).

Current models estimate hedgerow biomass carbon stocks to range from 5 t C ha<sup>-1</sup> to 45 t C ha<sup>-1</sup> (Falloon et al., 2004; Warner, 2011; Robertson et al., 2012) and SOC stocks from 43 t C ha<sup>-1</sup> to 136.8 t C ha<sup>-1</sup> (Falloon et al., 2004; Robertson et al., 2012). In the absence of measured data, such models use values extrapolated from average carbon stock values from other vegetation types. It may then be arguable that such assumptions, and therefore estimates, overlook the effects from carbon cycling processes unique to hedgerows.

#### 2.1.1 Assumed accumulation rates of SOC

In their study of UK field margins, Falloon et al. (2004) use expected SOC accumulation rates for natural woodland (1.17% yr<sup>-1</sup>) and arable conversions to pasture (1.30% yr<sup>-1</sup>) as representative values for tree and grass strips. Lacking data on carbon accumulation under hedgerows, Falloon et al. (2004) assume a rate between the values prescribed to grass margins and tree strips (1.23% yr<sup>-1</sup>). Given the limited availability of empirical data these assumptions seem justified. However, trees grown at a wide spacing have been found to demonstrate less apical dominance, forming larger crowns and heavier branches than trees grown in plantations (Williams et al., 1997). Such assumptions may have led to inaccuracies in their estimates. Nevertheless, Falloon et al. (2004) make it explicit that the values presented by the study are preliminary and based on hypothetical scenarios.

#### 2.1.2 Hedge structure and management effects

A number of factors determine SOC accumulation within woodlands: tree species, arrangement, age, and management. Although hedges may contain trees of a similar age and species to woodlands, their management regimes can differ dramatically. As hedgerow management practices affect hedge structure and dimensions (Baudry, 2000), it may be argued that differing management regimes result in carbon stock variation due to differing soil input characteristics.

The dominant pathway for carbon to enter the soil is through fine-root turnover (Al Afas et al., 2008; Upton and Burgess, 2013). Despite their relatively small contribution to overall root biomass, fine-root turnover accounts for up to 33% of annual NPP and responds rapidly to changes in the rooting environment (Jackson et al., 1997). Hedgerows in the UK are predominately maintained through flailing, where last year's growth is removed (Lofti et al., 2010; Benhamou et al., 2013). In most forest species the removal of above-ground biomass results in the reduction of fine-roots through necrosis (Jones et al., 1998; Jackson et al., 2000; Peter and Lehmann, 2000; Bayala et al., 2004; Crow and Houston, 2004; Montagnoli et al., 2012). Both repeated flailing and coppicing may therefore increase SOC through increased fine-root turnover and cuttings which are left behind. Unfortunately, the influence of management on carbon stocks and accumulation rates has not been widely explored, with most studies approaching hedge structure differently.

Falloon et al., (2004) and Warner (2011) calculate carbon stocks for hedgerows per 0.1m of hedge height, while Robertson et al. (2012) use three height classes (2m or less, >2m to 3m, >3m to 6m). Only Warner (2011) directly considers the effect of hedge management on carbon stocks by including laid hedges – laying being a traditional management style. All three studies, however, calculate SOC using hedge age, width and height as the sole influencing factors, overlooking potential differences in the frequency, quantity, and quality of soil inputs from different management practices.

Hedgerow trees are a common feature of European hedgerows, providing a number of ecosystem services (Forman and Baudry, 1984; Auclair and Dupraz, 1999; Barr, 2004; Ryszkowski and Kedziora, 2007). With large volumes of above-ground biomass they are likely to contribute significantly to the potential carbon storage of hedgerows (Wolton et al., 2014). It is however unclear whether any of the reviewed studies take into account the presence and abundance of these features.

### **2.1.3 Total carbon budgets**

Several studies indicate the importance of including the total carbon budget when assessing potential management options to increase the carbon sequestration potential of forestry (Pussinen et al., 1997; Deckmyn et al., 2004). Although the reviewed literature considers the potential carbon stocks and sequestration rates of hedgerows, total carbon budgets of the whole system are not included, with emissions from management activities, sequestration from durable products, and substitution of fossil fuels unaccounted.

Recent interest in managing hedgerows for woodfuel (Westaway et al., 2013; Wolton et al., 2014) raises the question: does coppicing, the cutting and removal of above-ground biomass, have a negative impact on SOC and is this balanced by the carbon savings in regrowth and substitution of fossil fuels? Total carbon budgets could be used to explore such questions. A simple carbon budget which captures the emissions from the management activities of coppicing, chipping and product transport and the substitution of fossil fuels was therefore included in the study.

## **2.2 Field research**

A limited number of studies have collected field data on the carbon storage of hedgerows. Prominent studies are that of Walter et al. (2003) who investigated the influence of contour hedging on SOC at a hill-slope scale and Follain et al. (2007) who considered the effects of hedgerow networks on SOC at a landscape scale. However, within these studies a number of methodological challenges are notable.

### **2.2.1 Sampling depths**

Most soil studies are limited to sampling depths of 20 to 30 cm (Nair, 2012). This depth is thought to correspond with the ploughed layer of arable land and is responsive to changes in management practices (Walter et al., 2003). However, a number of studies have demonstrated that although the establishment of trees on arable land can increase SOC within the upper soil horizon, it may also result in the depletion of soil carbon at depth (Jug et al., 1999; Upson and Burgess, 2013). For example, Upson and Burgess (2013) discovered a decline in SOC beneath poplar trees at greater soil depths than 60 cm. Inadequate sampling depths may therefore lead to the overestimation of carbon stocks within agroforestry systems. Sampling depths greater than 30 cm are therefore advised by Nair (2012) when considering tree-based systems.

Follain et al. (2007) and Walter et al. (2003) both deal with this methodological challenge well by using sampling depths greater than 30 cm. Follain et al. (2007) use a depth of 55cm although they concluded that a sampling depth of 30 cm accounts for 65% of total carbon stocks within a hedgerow network and allows for better comparison between studies.

Walter et al. (2003) take an arguably more robust approach to sampling by using the thickness of the A horizon (i.e. the surface soil horizon where organic matter is mixed with mineral matter) of the soil to determine sampling depth. Samples were taken in 10 cm increments for the whole of the A horizon and the first 10 cm of the mineral horizon (Walter et al., 2003). Walter et al. (2003) found soil depth to be the single most significant factor in SOC content variation. Under the hedges, SOC content of the whole A horizon was systematically greater than that of the top 30 cm, implying that a sampling depth of 30 cm would have likely been inadequate.

### 2.2.2 Soil analysis

The residence time of SOC is a function of aggregate size (Christensen, 2001; Elliott, 1986; Parton, 1987; Six et al., 2000, 2002a, 2002b). Macroaggregates (250-2000  $\mu\text{m}$ ), microaggregates (53-250  $\mu\text{m}$ ), and silt and clay (<53  $\mu\text{m}$ ) soil fractions can have a mean residence time of 1 to 10, 25, and 100 to 1000 years respectively (Howlett et al., 2011). The formation of microaggregates and silt and clay aggregates within macroaggregates is preferential for longer-term carbon storage, due to carbon within smaller fractions being protected from microbial activity (Jastrow et al., 1998; Howlett et al., 2011). Determining the carbon content of different soil fractions therefore allows for the estimation of SOC residence times within a given soil and the potential for long-term carbon storage (Howlett et al., 2011). Walter et al. (2003) and Follain et al. (2007) measure SOC for the whole soil rather than different soil fractions. The location and residence time of carbon within the soils therefore remains undetermined.

Both Walter et al. (2003) and Follain et al. (2007) use soil samples ground to 2 mm for analysis. Carbon is also likely to be stored within larger soil fractions (Howlett et al., 2011; Nair, 2012). Exclusion of these larger fractions may therefore have resulted in an underestimation of SOC.

Both Walter et al. (2003) and Follain et al. (2007) use a single sampling event, overlooking any temporal variation effects in carbon stocks. Within tree-based systems changes in carbon stock are unlikely to be linear (Nair, 2012) and given the slow response and long residence times of SOC (Howlett et al., 2011) a chronosequential study would be required to determine changes in hedgerow carbon stocks over time.

### 2.2.3 Scaling-up

Walter et al. (2003) attempt to extrapolate their field survey results to a landscape scale considering different hedge network densities. Follain et al. (2007) criticise Walter et al. (2003) for their generalization of local carbon stocks at a landscape scale. Follain et al. (2007) argue that favourable study site conditions with hedgerows positioned perpendicular to slope, disregard for the high variability of SOC stocks at a landscape scale, and a lack of information on the real distribution of hedges acting as barriers within the network, are likely to have resulted in the overestimation of carbon stocks at a landscape scale.

## 2.3 Conclusion

The validity of current estimates for hedgerow carbon stocks and accumulation rates is limited by incomplete information on the effect of vegetation type, hedge structure, management practices, total carbon budgets, and the landscape-scale impact of local carbon storage processes. Furthermore, methodological challenges in SOC analysis restrict the accuracy of data regarding the location and form of carbon within soils. The reviewed studies are however explicit about their limitations, and given the absence of empirical data, present a useful starting point for future enquiry. Collection of further empirical data on the carbon sequestration potential of hedgerows is needed to validate existing predictions and models and to inform decisions not only at a farm management level but also for wider policy.



### 3. Methods

In order to estimate carbon stores, develop model adaptations, and calculate carbon budgets, data was collected from three hedges on a livestock farm in the South of England. Methods included quantification of the biomass productivity of each hedge, current SOC stocks, leaf litter production and measurement of coppice regrowth. Due to the absence of replicates, no statistical analyses were carried out within this study.

#### Site description

Elm Farm is an 85 hectare organic livestock farm in the South East of England. The farm has an average annual rainfall of 71cm. The soil type is mainly Wickham Series clay, poorly drained clay loams susceptible to structural damage. The hedges on Elm Farm have not been actively managed for a number of years, aside from occasional side flailing to maintain field sizes and statutory roadside management. Results from a survey of all hedges on the farm carried out in July 2013 showed that the dominant woody species is Blackthorn (*Prunus spinosa*), with other commonly recorded species being Hawthorn (*Crataegus monogyna*), Hazel (*Corylus avellana*), Pussy Willow (*Salix caprea/cinerea*) and Oak (*Quercus robur*). Blackthorn, bramble and rose outgrowth is also common, resulting in wide unruly hedges, often with the existing fences being engulfed by this shrubby outgrowth.

#### Experimental design

The experimental design consisted of paired 15m cut and uncut plots (Figure 1) established in three different hedgerow types: blackthorn dominated, hawthorn dominated, and hazel dominated.

These hedges were chosen based on the following factors:

- The management history of the hedgerow and the stage in the hedgerow management cycle (Hedgelinek, 2008). Hedges at a suitable stage for coppice management were selected.
- The ability to identify two 15m plots within the hedges broadly comparable in terms of woody species composition and hedgerow structure.
- The ability to site plots at least 5m away from hedge ends and intersections with other hedges and mature tree canopies.
- Ease of access to the hedge for management activities.
- Boundary and roadside hedges were excluded due to potential management conflicts.

One 15m plot from each hedge type was chosen at random and all woody material coppiced by hand. Any outgrowth of bushy vegetation from the hedge was cut back prior to coppicing.

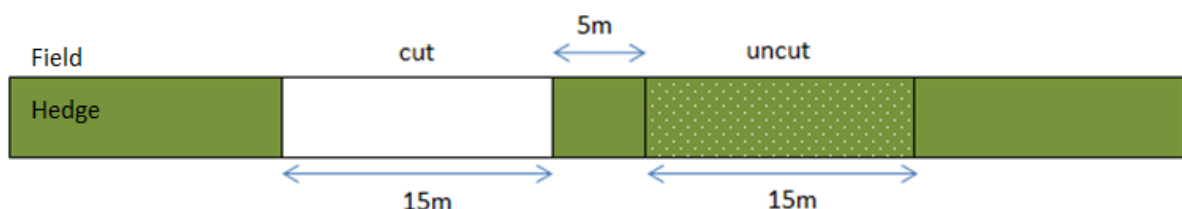


Figure 1. Trial plot set up: a 15m cut plot next to a 15m uncut plot with a 5m uncut buffer between.

## Definition of the system boundaries

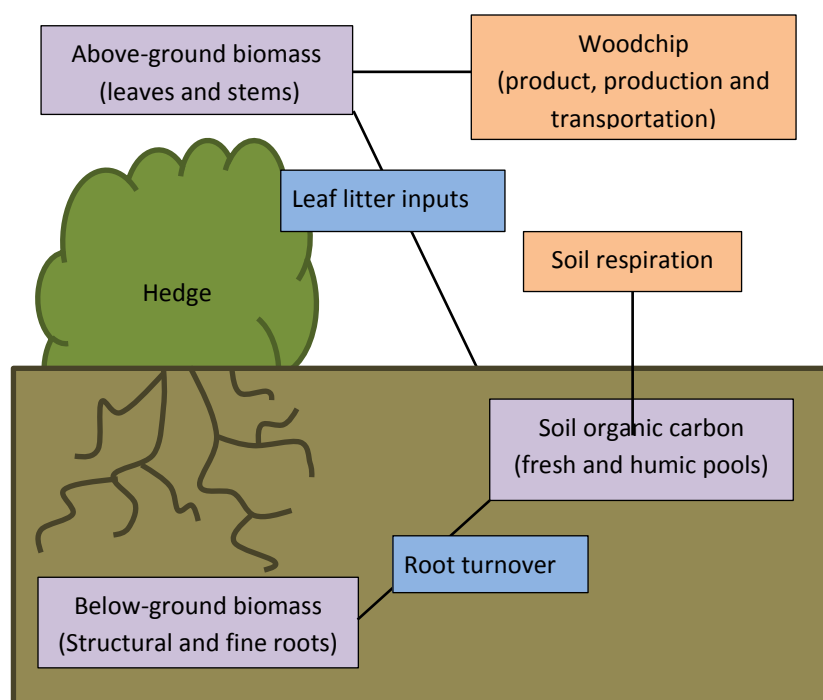
For the estimation of carbon stores, model scenarios and carbon budgets, two hedge scenarios (associated with cut and uncut plots) were used and referred to as ‘unmanaged’ and ‘managed’; unmanaged referring to hedges occasionally flailed to control outgrowth and not managed by coppicing, and managed referring to hedges managed on a 15 year coppice rotation for woodfuel.

### *Managed hedges:*

The carbon stores and flows associated with the managed hedge system are depicted in Figure 2. There are assumed to be six main carbon pools within the hedgerow system: two within the above-ground biomass (leaves and stems); two within the below-ground biomass (structural roots and fine roots); and two soil carbon pools (fresh soil carbon and humic soil carbon). Carbon flows between these pools include leaf-litter from the above-ground biomass of the hedge and below-ground fine root turnover. Carbon flows out of the system include soil respiration and woodchip produced from above-ground biomass. Although the woodchip produced substitutes the use of carbon from fossil fuels for energy production, it is burnt and therefore does not store carbon in the long-term. Carbon outputs also consist of direct fossil fuel combustion from fuel used in harvesting and processing machinery, the transportation of the woodchip and the embedded energy of the machinery itself. Energy inputs related to labour were not considered within this study.

### *Unmanaged hedges:*

The carbon stores and flows within the system associated with the unmanaged hedge system are identical to those of the managed system. Carbon flows out of the unmanaged system however exclude those from the production of woodchip and the woodchip itself.



**Figure 2. Schematic diagram of carbon stores (purple), carbon flows within the system (blue), and carbon flows out of the system (orange) under the managed hedge system. Woodchip (product, production and transportation) is excluded under the unmanaged scenario.**

### 3.1 Estimation of current carbon stores

In order to estimate the current carbon stores of both managed (cut) and unmanaged (uncut) hedges the following parameters were measured: biomass harvested following coppicing, coppice regrowth, leaf litter production and soil organic carbon.

#### 3.1.1 Above-ground biomass

Pre-harvest both the average width and height of each hedge plot was estimated using a 2m long pole. These dimensions were later used to estimate the biomass productivity of the regrowth following coppicing. Post-harvest all woody material coppiced from each managed section was chipped using a forestry chipper, transported to a hard standing and transferred into 1 tonne dumpy bags and weighed using a tractor-mounted spring balance.

Moisture content (MC) of the chip was specified as a percentage of the total sample weight and was determined using a simple oven drying method, where five representative samples of approximately 0.25kg each were taken from each chip pile, weighed (green weight) and dried in an oven at 100 degrees Celsius until a constant mass was reached (dry weight). The moisture content was then calculated by subtracting the dry weight from the green weight in order to calculate the weight of water. The weight of water was then divided by the green weight to calculate the moisture content of the sample. The average moisture content of the five samples was taken as the average moisture content of the whole chip pile. The dry mass (0% MC) of woodchip produced per metre of hedge was then calculated and a carbon content of 0.49 was assumed for the fraction of C within the coppiced biomass (Matthews 1993). These figures were then used to estimate the above-ground carbon store of each unmanaged hedge scenario. Carbon stored within the unutilised coppice stools remaining after coppicing was not measured due to the difficulties of stool excavation. Figures for total carbon stored above-ground may therefore be underestimates.

To determine the above-ground carbon store of the managed hedge scenarios the number of living shoots present on each stool at the end of May 2014 was recorded. For the first year following coppicing re-growth measurements are taken at two-monthly intervals throughout the growing season. As per Croxton et al. (2003) the five longest shoots from each live stool were measured and an average per stool calculated. At the end of the growing season, November 2014, the total number of shoots on each stool was re-counted along with the length of the five longest shoots and average width of the hedge to give the total growth in the first year following coppicing. Due to the suckering nature of blackthorn, root sucker regrowth was recorded as associated with a stool if it occurred within 20cm of the stool.

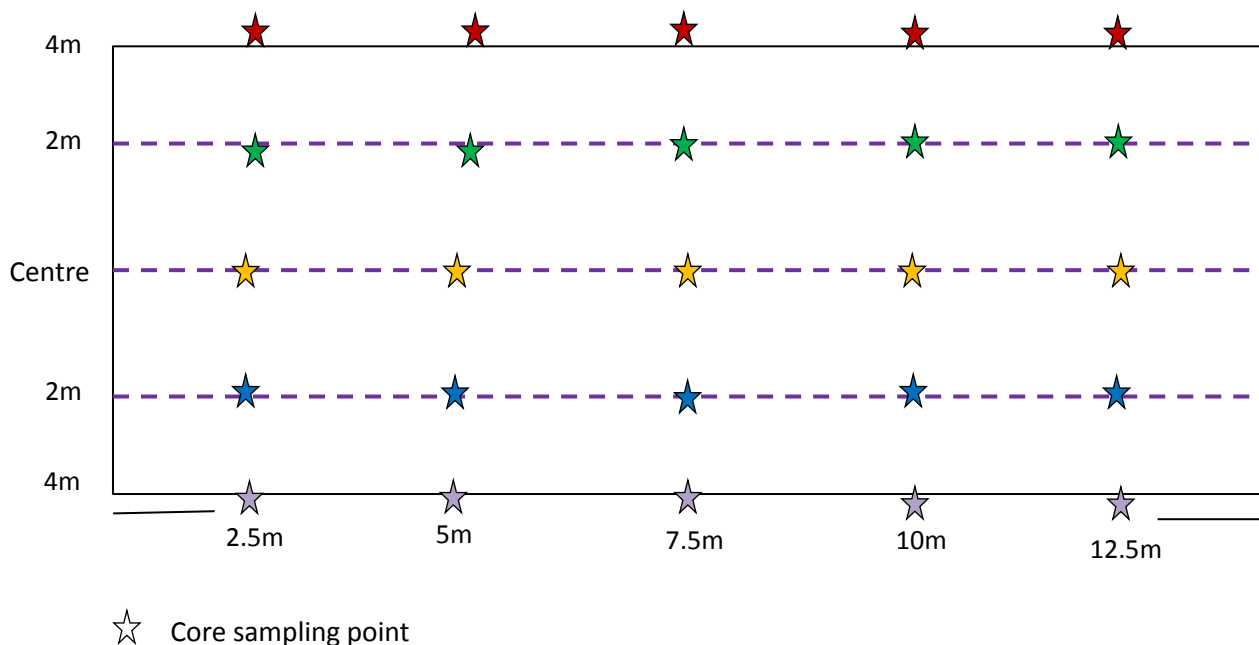
In order to estimate the total carbon stored within the regrowth of each coppiced hedge plot the following non-destructive method was used. By multiplying the average height by the average width of the regrowth, the volume of regrowth per metre was estimated. The volume of the regrowth per metre was then divided by the volume per metre of the hedge plot before coppicing to provide a scaling factor. This scaling factor was then multiplied by the dry mass of the woodchip produced per metre of hedge when coppiced. Total carbon within the regrowth material was also assumed to be 0.49 of the estimated biomass (Matthews 1993). In effect, the biomass production of the mature hedge was used to estimate that of the coppice regrowth. This method does, however, assume the regrowth to be of a similar density and composition to that of a mature hedge. Regrowth is, however, unlikely to have contained the same quantities of woody material and to have been as dense as the mature hedge. These above-ground carbon stores are therefore treated as overestimates.

### 3.1.2 Soil organic carbon

Total soil organic carbon (SOC) was determined for both cut and uncut hedge plots using a composite sampling design one year after coppicing. The distribution of SOC within different soil carbon pools (e.g. fresh and humic) was not determined due to the high resource requirement of fractionation and analysis.

#### Composite sampling design

A composite sampling design was chosen where five soil cores were taken along five transects running parallel to the hedge (Figure 3). One transect ran as close to the centre of the hedge as possible, and the other transects at 2m and 4m from the centre of the hedge on either side. A total of 25 cores per hedge plot were taken. Each soil core was then divided into four layers determined by depth (0-7.5, 7.5-15, 15-30, 30-50cm). The same layer from each soil core within a transect was then combined to form a composite sample for that depth. Using composite sampling substantially reduces the number of samples required for carbon analysis without significant loss in the precision of mean estimates. Samples were then oven-dried at approximately 40°C. A temperature of 40°C was used as temperatures above 65°C can activate carbon oxidation.



**Figure 3.** An aerial view of plot layout. Location of each core sample represented by a star. Each colour indicates a different transect composed of five sampling points i.e. yellow stars represent the centre transect.

Samples were then sent to Chemtec Environmental Ltd. ([www.chemtech-env.co.uk](http://www.chemtech-env.co.uk)) for analysis where total organic carbon (TOC) was determined using the following methodology:

A representative sub-sample was crushed using a soil mill. The whole of each sample was crushed and no stones were removed (if any were present). Approximately 0.200g of sample was weighed into a filtering crucible. The crucible with sample was then placed into a wash block and a vacuum applied. Hydrochloric acid (10%) was then added to the crucible in small aliquots until effervescence stopped. Once effervescence stopped, the crucible was filled with hydrochloric acid (10%) and allowed to drain. Hydrochloric acid was again added to the crucible and again allowed to drain. The

remaining sample was then washed three times with water with the crucible being allowed to drain between each washing. The crucible was then placed in an oven to dry at  $105 \pm 5^\circ\text{C}$  for at least 2 hours. Iron chip and tungsten accelerator were then added to the sample which was then analysed using an Eltra CS-800 Carbon Analyser.

### **Sampling depth**

Sampling depths 0-7.5cm and 7.5-15cm were selected for compatibility with the UK's Countryside Survey (Countryside Survey, 2007) and the layer 15-30cm was chosen based on the IPCC's recommendation of 30cm as a default sampling depth (IPCC, 2003). The depth of 30cm corresponds with that of the ploughed layer (Walter et al., 2003) and has been shown to account for 65% (Follain et al., 2007) and 78.8% (Howlett et al., 2011) of total carbon stock within hedgerow networks. Although establishment of trees on arable land can lead to increased SOC within the upper soil horizon it may also result in the depletion of soil carbon at depth (Vesterdal and Ritter, 2002; Jug et al., 1999; Ritter et al., 1999; Upson and Burgess, 2013). An inadequate sampling depth may therefore lead to the overestimations of carbon stocks. In an attempt to address potential overestimation while taking into account practicality, the final sampling depth of 30-50cm was chosen.

### **Timing**

In order to reduce temporal variations in SOC, soil sampling was undertaken in late January when biological activity is low (Vladimir et al., 2005).

### **Bulk density**

Bulk density is the weight of soil in a given volume and is required for the calculation of soil carbon stocks. Since bulk density is generally less variable than SOC (Aynekulu et al., 2011) it was determined by taking three randomly chosen sampling sites per hedge plot (cut and uncut). In each location a 50cm deep pit was dug and two bulk density cores taken from the side wall using a cylinder of known volume; one core taken from the 0-25cm horizon, and one between 25-50cm. Bulk density samples were then oven-dried and the dry weight divided by the volume of the cylinder to give the bulk density of the soil. The average bulk density of samples taken from the upper horizon of each plot was later applied to the results of SOC samples taken at 0-7.5cm, 7.5-15cm, and 15-30. The average bulk density of samples taken in the lower horizon was applied to SOC samples taken at 30-50cm.

### **3.1.3 Below-ground carbon**

Carbon stored within hedge roots was not measured directly due to the difficulties of root excavation. Instead below-ground carbon stores were estimated assuming 0.25 of the total net carbon assimilated each year is allocated to root growth (Grogan and Matthews, 2002). Below-ground carbon stores were therefore calculated as a third of the above-ground carbon store (above ground is 0.75 and below ground is 0.25 therefore to estimate the below ground store from the above ground store divide by  $75/25 = 3$ ).

Although coppicing is a well-known silvicultural practice, relatively little is known about root-system responses to coppicing (Dickman et al., 1996). The response of fine-roots to coppicing has, however, been found to vary with tree species (Crow et al., 2004), stool age, and coppice rotation length (Bedeneau et al., 1989). Coppicing may also indirectly affect fine-root turnover due to increased soil temperatures following canopy removal (Montagnoli et al., 2012). Although it is expected that coppicing a tree will cause a short-term decline in fine-root biomass this may not always be the case. For example, certain poplar clones have been shown to produce substantial fine-root production in the spring following coppicing with no signs of significant root dieback (Dickman et al., 1996). Such findings indicate that the root systems of certain trees contain adequate carbon and nitrogen reserves to induce a flush of root-growth despite removal of above-ground biomass (Dickman et al.,

1996). Additionally, Bedeneau et al. (1989) and Crow et al. (2004) suggest that longer coppice rotations, and hence increased stool age, equate to larger carbon and nitrogen reserves within roots.

Although it is assumed that coppicing was followed by dieback of fine roots within the three hedge plots, a conservative figure for the proportion of below-ground biomass lost after coppicing was used as, in most tree species coppicing leads to dieback of fine roots followed by rapid recovery of their biomass (Montagnoli et al., 2012), and due to the small contribution of fine roots to overall root biomass (Jackson et al., 1997). When calculating the below ground biomass of the recently coppiced hedge only 10% of the roots are assumed to have died back so as to not over-estimate root necrosis.

#### **3.1.4 Estimation of carbon flows**

To determine the potential carbon entering the soil through the foliage pool, leaf litter samples were collected from each plot after leaf fall in December and dry mass determined (Verlinden et al., 2013). For each plot (managed and unmanaged) a 25cm wide transect was established through the centre of the plot perpendicular to the hedge. Samples using a 1m x 0.25m quadrat were taken every 1m along each transect with all leaf litter removed and weighed and a sub-sample dried. Although different plant tissue types vary in carbon content (Matthews et al., 1994), a leaf litter carbon content of  $0.4 \text{ g C (g DM)}^{-1}$  (Grogan and Matthews, 2002) was assumed for simplicity in the absence of data on the production of different components within the leaf litter (twigs and leaves) and their carbon content. This figure is lower than that used to estimate the carbon in the above-ground biomass which is likely to contain larger quantities of woody material.

Woodchip production is considered a carbon flow out of the system. See section on above-ground biomass estimation for details on its calculation.

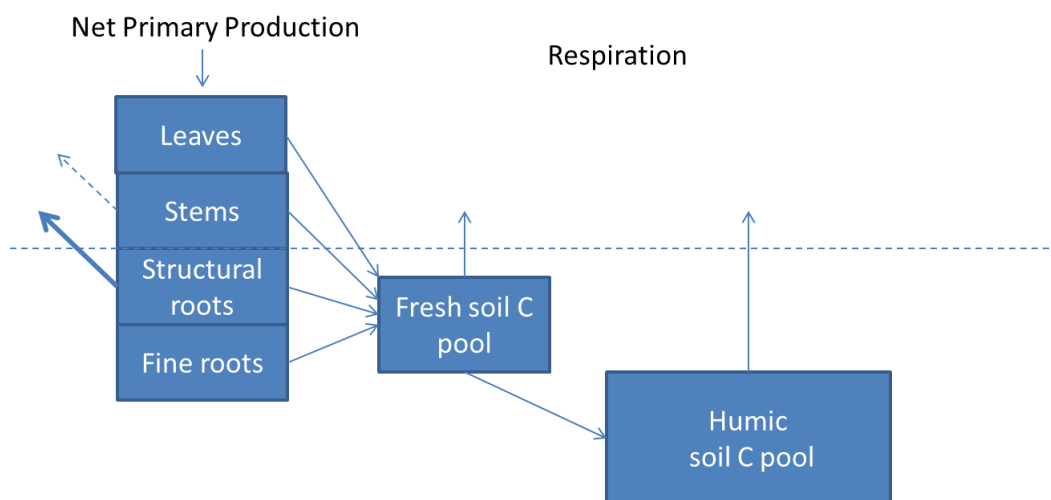
## 3.2 Modelling

Given the limited understanding of, and data on, carbon dynamics under hedgerow systems, the aim was to develop a model that could be used as a starting point for future enquiry. A process-based model developed by Grogan and Matthews (2002) and described in their paper 'A modelling analysis of the potential for soil carbon sequestration under short rotation coppice willow bioenergy plantations' was adapted using key parameters specific to both managed and unmanaged scenarios for each hedge type (blackthorn dominated, hawthorn dominated and hazel dominated).

### 3.2.1 Model Description

The model developed by Grogan and Matthews (2002) aims to analyse the potential for soil carbon sequestration under short rotation coppice (SRC) willow. The model is based on a simple mass balance of the major pools and fluxes of carbon within a managed woodland and SRC plantation and was adapted from the carbon cycle modelling structure used in CENTURY (Parton et al. 1993) and RothC (Coleman & Jenkinson, 1996). Data from the Greescroft Wilderness site, a naturally regenerated woodland in South England, was used to develop and calibrate the model, then key parameters specific to SRC willow in the UK, such as production data and management practices, were applied to the model in order to estimate both above and below-ground carbon pools under SRC.

Their model assumes there to be six main carbon pools (Figure 4): two within the above-ground biomass (leaves and stems); two within the below-ground biomass (structural roots and fine roots); and two soil carbon pools (fresh soil carbon and humic soil carbon).



**Figure 4. The main carbon pools and fluxes used in the carbon sequestration model developed by Grogan and Matthews (2002). The arrows to the left of the diagram indicate the carbon losses due to the short rotation coppice harvest (dotted line) and plantation reestablishment (solid line). (Diagram adapted from Grogan and Matthews, 2002).**

Grogan and Matthews (2002) characterise the major fluxes and pools as follows:

**Biomass production:**

$$\frac{dW}{dt} = S e_r (1 - e^{-kL}) f_c \times 10$$

Here Grogan and Matthews use a version of Beer's Law where  $dW/dt$  is the growth rate of tree biomass (above- and below-ground) expressed as units of carbon ( $\text{kg C ha}^{-1} \text{ yr}^{-1}$ ),  $S$  is the annual receipt of short-wave solar radiation ( $\text{MJ m}^{-2} \text{ yr}^{-1}$ ),  $e_r$  is the efficiency of conversion of this solar radiation into biomass ( $\text{g DM MJ}^{-1}$ ),  $k$  is the light extinction coefficient,  $L$  is the leaf area index (leaf), and  $f_c$  is the fraction of carbon in the biomass ( $\text{g C (g DM)}^{-1}$ ). The factor of 10 is used to convert  $\text{g C m}^{-2}$  into  $\text{kg C ha}^{-1}$ . Their model is intended for investigating carbon changes over a century and runs on a yearly time-step.

Despite a preliminary study carried out by Matthews et al. (1994) indicating that different tissue types vary in carbon content (stem wood =  $0.38 \text{ g C (g DM)}^{-1}$  and fine roots =  $0.46 \text{ g C (g DM)}^{-1}$ ), Grogan and Matthews use a value of  $f = 0.4 \text{ g C (g DM)}^{-1}$ . This is done for simplicity in the absence of any data on biomass and production of different components and their carbon content. Grogan and Matthews assume a value of  $k = 0.6$  for the light extinction coefficient under both woodland and SRC scenarios although they provide no justification.

Of the total net carbon assimilated each year, Grogan and Matthews assume that a fixed proportion ( $f_r$ ) is allocated to new root growth, with the remainder (i.e.  $1 - f_r$ ) being allocated to above-ground growth. A value of 0.25 is assumed for  $f_r$ , which is thought to be consistent with figures reported by Lambers et al. (1998). A value of  $S = 4000 \text{ MJ m}^{-2} \text{ yr}^{-1}$  is used based on typical values measured at Silsoe which is located 50km away from Greescroft.

Based on measured mean annual production data for a SRC site in the UK ( $14600 \text{ kg DM ha}^{-1} \text{ yr}^{-1}$ ) a value of  $e_r = 0.67 \text{ g MJ}^{-1}$  is used (Armstrong, 2000). For the Greescroft Wilderness site a value of  $e_r = 0.21 \text{ g MJ}^{-1}$  is used based on observed production data (Jenkinson et al. 1992). The higher value for the SRC reflects not only the faster growth rate in the SRC system but also the higher planting densities.

**Leaf area index:**

Grogan and Matthews use Leaf area index ( $L$ ) as one of the main variables from which annual biomass production is derived and it is also used in their model to determine the amount of leaf litter reaching the soil.

For Greescroft, Grogan and Matthews assume leaf area index to increase linearly to a maximum of  $9 \text{ m}^2 \text{ m}^{-2}$  after 10 years and that it remains at this value thereafter. This value is chosen based on the leaf litter data from the Greescroft Wilderness site (Poulton, 1996) and  $L$  values reported for temperate deciduous forests (Larcher, 1995).

For the SRC,  $L$  is based on measured production data (Cannell et al. 1987) and expressed as a function of age since planting ( $L = 2.3$  in year 1;  $4.6$  in year 2; and  $7$  in the third and subsequent years).



### Carbon inputs from canopy and root system:

Grogan and Matthews assume all leaf material produced in a given year falls at the end of that year, and enters the 'fresh' soil carbon pool and that there is an age dependent amount of wood detritus that falls to the ground and enters the fresh carbon pool. They include not only branches and twigs but also dead trees from self-thinning. In their model the annual amount of carbon input into the 'fresh' soil carbon pool due to canopy sources under both scenarios is calculated as:

$$W_{Cin} = \frac{L f_c}{SLA} \times 10^5 + W_{AG} f_w$$

Where SLA is the specific leaf area, and  $f_w$  is the fraction of above-ground carbon input ( $W_{AG}$ , kg C ha<sup>-1</sup>), that enters the fresh carbon pool as woody material rather than leaf litter.

This fraction is determined as  $f_w = 0.015/(1+100e^{-t})$ . For Greescroft  $t$  is the number of years since the start of the simulation; for the SRC  $t$  is the number of years since the last coppicing. This results in almost all of the carbon input coming from the canopy in the early years of growth and, as the stand ages, progressively more woody material starts to enter the fresh carbon pool up to a maximum of 1.5% per year of the above-ground biomass. However it is noted by Grogan and Matthews that in reality very little woody biomass would enter the soil under a SRC system as the woody biomass is removed before it becomes old enough to fall.

They assume that a proportion ( $f_{FRTO}$ ) of the newly assimilated carbon allocated to the root system each year is lost in fine root turnover and enters the fresh carbon pool; this value also includes carbon lost through root respiration and rhizodeposition. Grogan and Matthews assume that a fraction of the plant carbon below-ground also enters the fresh carbon pool through the death and decay of woody root biomass. They assume this fraction to be the same as that calculated for the above-ground wood detritus. The total carbon inputs into the fresh soil carbon pool from the root system ( $W_{Rin}$ , kg C ha<sup>-1</sup>), are calculated as:

$$W_{Rin} = \frac{dW}{dt} f_r f_{FRTO} + W_{BG} f_w$$

Where  $W_{BG}$  is the weight of carbon below-ground in the root system (kg C ha<sup>-1</sup>). It is assumed that 50% of the below-ground carbon in the root system is lost to fine root turnover on an annual basis ( $f_{FRTO} = 0.5$ ) for Greescroft. However a higher value is assumed for the SRC ( $f_{FRTO} = 0.85$ ) due to increased root necrosis following frequent coppicing of above-ground biomass.

### Decomposition of soil carbon pools:

In order to keep their model simple Grogan and Matthews (2002) assume that there are two organic carbon pools in the soil, that of fresh carbon originating from recent tree growth, either from the canopy or from the root system (fresh organic matter: 'FOM'), and that of long-lived soil organic carbon (or humus: 'HUM'). The decay of these two pools was assumed to follow first-order kinetics with different rate constants as calculated below:

$$\frac{dC_p}{dt} = -C_p k_p$$

Where  $dC_p/dt$  is the rate of change of the carbon pool  $p$  in question (kg Cha<sup>-1</sup> yr<sup>-1</sup>),  $C_p$  is the size of the pool (kg C ha<sup>-1</sup>), and  $k_p$  is the rate constant for each pool (yr<sup>-1</sup>). Each year, decomposition losses of carbon are subtracted from the pool in question, and any remaining carbon in the fresh organic pool is assumed to enter the humic carbon pool.

Through preliminary simulations Grogan and Matthews found their model's accuracy to be particularly sensitive to rates of decay. To gain the most realistic rates of decay for the two carbon pools the model rate constants ( $k_{\text{FOM}}$  and  $k_{\text{HUM}}$ ) of decay were calibrated with the Greescroft Wilderness soil carbon data. The two rate constants that gave the best fit with the Greescroft data were  $0.786 \text{ yr}^{-1}$  for  $k_{\text{FOM}}$  and  $0.0031 \text{ yr}^{-1}$  for  $k_{\text{HUM}}$  corresponding to turnover times of around 1.27 years for the fresh carbon pool and 325 years for the humic carbon pool.

### **Coppice rotation:**

The SRC is harvested at intervals of three years and is modelled by removing all of the above-ground biomass from the system while the root biomass is assumed to remain in the soil. Plantation re-establishment is simulated by Grogan and Matthews by removing both above and below-ground biomass every 24 years.

### **Soil carbon distribution:**

Grogan and Matthews assume that SRC plantations are able to influence carbon sequestration to a maximum depth of 0.5 m and that 80% of the humic pool occurs in the top 0.23 m soil layer.

### 3.2.2 Hedge specific adaptations

In order to adapt the model developed by Grogan and Matthews to a hedgerow scenario, their original model was replicated in Python (3.4) using the equations and parameters stated in their paper. Both the original model outputs produced by Grogan and Matthews and the model's replication can be seen in Figure 5-8. Despite use of the details provided by Grogan and Matthews, the replicated model was unable to produce the same predictions for changes in soil carbon. As stated by Grogan and Matthews, their modelling approach is relatively simple and relies on a number of assumptions, resulting in a model which is highly sensitive to inputs and parameter values. Even small variations in the degree of precision at which simulations are run may therefore have resulted in these differences (e.g. values stated in their paper rounded to two decimal places).

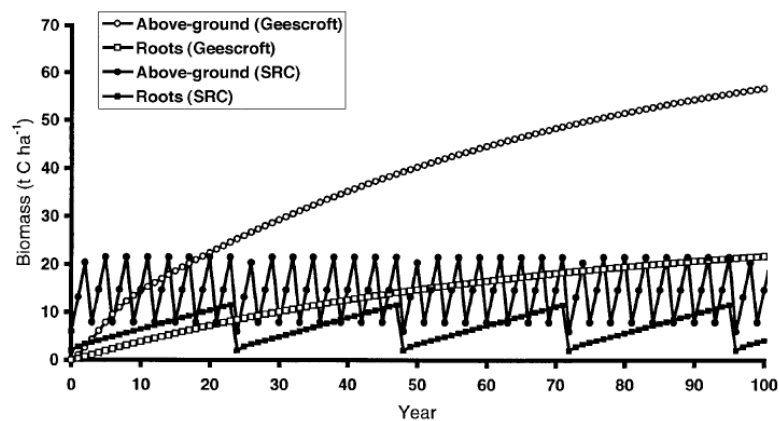


Figure 5. The predicted above and below-ground biomasses for the Geescroft Wilderness site since regeneration and for the SRC willow plantation system over 100 years as presented by Grogan and Matthews (2002) in their paper. SRC harvest interval is three years and total removal for replanting occurs every 24 years.

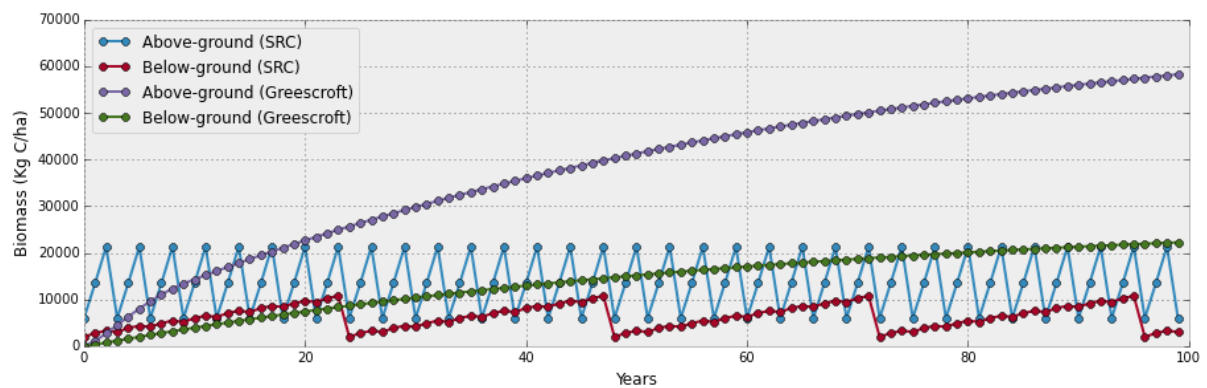
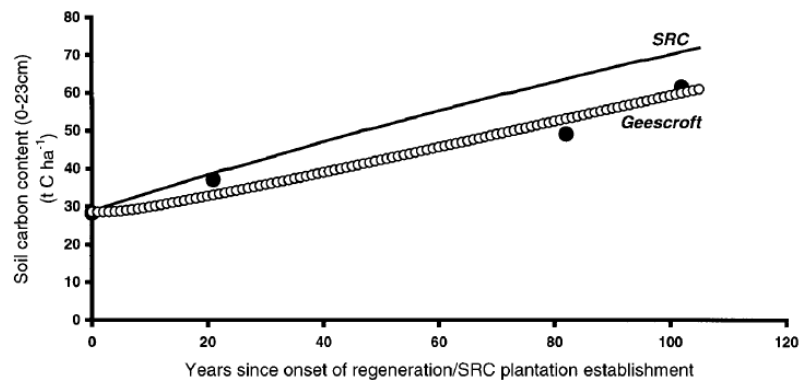
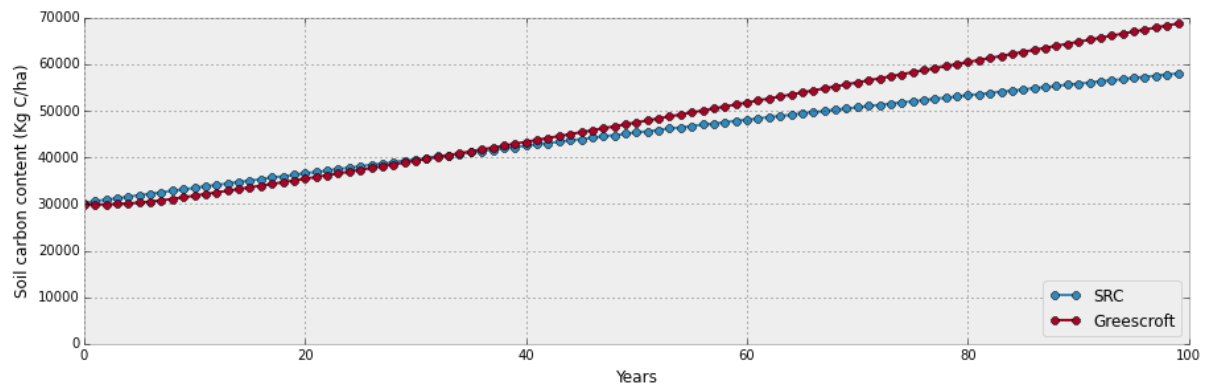


Figure 6. Replication using Python (3.4) of the predicted above and below-ground biomasses for the Geescroft Wilderness site since regeneration and for the SRC willow plantation system over 100 years using the model developed by Grogan and Matthews (2002).



**Figure 7.** The predicted changes in soil carbon in the 0-23cm layer for the Geescroft Wilderness woodland and for SRC growing on the same site as presented by Grogan and Matthews (2002) in their paper. SRC harvest interval is three years and total removal for replanting occurs every 24 years.



**Figure 8.** Replication using Python (3.4) of the predicted changes in soil carbon in the 0-23cm layer for the Geescroft Wilderness woodland and for SRC growing on the same site using the model developed by Grogan and Matthews (2002).

Key parameters specific to the three hedge types were then applied to the model in order to predict both above- and below-ground carbon pools over 100 years. For each hedge species (hazel, hawthorn and blackthorn) carbon pools were calculated for both managed and unmanaged scenarios.

The following parameters were adapted to model the hedge specific scenarios; all other parameters remained the same as in Grogan and Matthews' original model.

#### **Biomass production:**

Values for the conversion efficiency of solar radiation into biomass ( $e_r$ ) were calculated for each scenario as:

$$e_r = \left(\frac{B}{S}\right) + \left(\frac{f_r B}{S}\right)$$

Where  $e_r$  is the efficiency of conversion of this solar radiation into biomass ( $\text{g DM MJ}^{-1}$ ),  $B$  is the annual above-ground biomass production ( $\text{g DM m}^{-1} \text{yr}^{-1}$ ),  $S$  is the annual receipt of short-wave solar radiation ( $\text{MJ m}^{-2} \text{yr}^{-1}$ ), and  $f_r$  is the fraction of the total net carbon assimilated each year allocated to new root growth, with the remainder (i.e.  $1-f_r$ ) being allocated to above-ground growth.

The annual above-ground biomass production ( $B$ ) for the unmanaged scenarios was calculated by dividing the above-ground biomass production per metre by the estimated age of the hedge. The age of each of the three hedges had been estimated by counting the number of annual rings present on three randomly chosen stumps shortly after coppicing. For the managed scenarios,  $B$  is the estimated above-ground biomass production per metre one year after coppicing.

A value of  $4000 \text{ MJ m}^{-2} \text{yr}^{-1}$  was used for  $S$ , the annual receipt of short-wave radiation, the same figure as used by Grogan and Matthews. This figure is based on typical values measured at Silsoe located 50km from Greencroft. Silsoe is approximately 90km north east of Elm Farm, Newbury (the location of the three hedges). This  $S$  value was therefore thought adequate for the hedgerow scenario model. In the absence of hedge specific data, the same  $f_r$  value (0.25) as used by Grogan and Matthews was used.

The  $e_r$  values calculated for each scenario are shown in Table 1. The higher  $e_r$  values for the hazel and hawthorn managed scenarios reflects the faster growth rate following biomass harvest due to the invigorating effects of coppicing (Dickman et al., 1996). In the blackthorn experimental plot, regrowth following coppicing was poor, while in contrast, hazel was found to respond exceptionally well to coppicing.

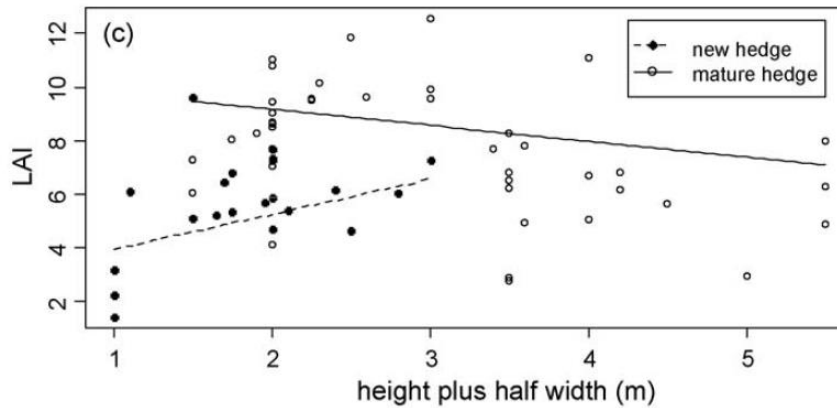
**Table 1. Conversion efficiency of solar radiation into biomass ( $e_r$ ) values calculated for each hedge scenario based on estimated above- and below-ground biomass production and an annual receipt of short-wave radiation of  $4000 \text{ MJ m}^{-2} \text{yr}^{-1}$ .**

Hedge scenario	Conversion efficiency ( $e_r$ )
<b>Blackthorn</b>	
Managed	0.47
Unmanaged	1.18
<b>Hawthorn</b>	
Managed	0.56
Unmanaged	0.48
<b>Hazel</b>	
Managed	1.61
Unmanaged	0.48

### Leaf area index:

Leaf area index (LAI) values for both managed and unmanaged hedge scenarios were taken from Pocock et al. (2010) who developed a predictive model for hedgerow LAI based on measured data. Variation in hedge LAI was found to be a function of hedge height and width (Figure 9).

Although not species specific, the average measured LAI for a mature hedgerow excluding basal vegetation was  $6.8 \text{ m}^2/\text{m}^2$ . Under the unmanaged scenarios LAI is therefore assumed to have increased linearly up to a maximum of  $6.8 \text{ m}^2/\text{m}^2$  after ten years (Table 2). In the absence of empirical data, this time scale is based on the assumption that most hedgerows reach maturity within this period.



**Figure 9. Model presented by Pocock et al. (2010) depicting the relationship between hedge size and LAI.**

For the managed scenarios LAI values (Table 2) were based on the linear progression in LAI as a function of hedge size for new hedges as shown in Figure 9. For each metre gained in hedge size LAI increases by a factor of one. Once again LAI was assumed to increase linearly to a maximum of  $6.8 \text{ m}^2/\text{m}^2$ .

**Table 2. Leaf area index (LAI) values for both managed and unmanaged hedge scenarios based on measured data and predictive LAI model by Pocock et al. (2010).**

Annual LAI values (until maximum is reached)				
Year	Unmanaged (all species)	Blackthorn	Hawthorn	Hazel
1	0.62	0.74	1.09	2.62
2	1.24	1.48	2.18	5.24
3	1.82	2.22	3.27	6.80
4	2.48	2.96	4.36	
5	3.10	3.70	5.45	
6	3.72	4.44	6.54	
7	4.34	5.18	6.80	
8	4.96	5.92		
9	5.58	6.66		
10	6.80	6.80		

### **Carbon inputs from canopy and root system:**

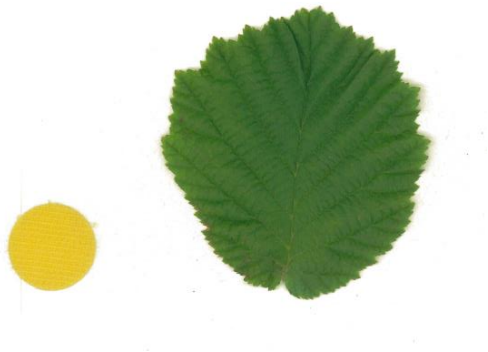
Specific leaf area (SLA) is the ratio of leaf area to dry mass (Garnier et al., 2001) and is used by Grogan and Matthews to calculate the above-ground carbon input that enters the fresh carbon pool.

Values for SLA were determined for each hedge species from six leaf samples in July 2015 (Table 3). The area of each leaf was measured using ImageJ ([www.imagej.net](http://www.imagej.net)), a digital image analysis software designed for scientific multidimensional images (Bakr, 2005; O'Neal et al., 2002).

**Table 3. Specific leaf area values used for each hedge species.**

Hedge species	SLA (cm <sup>2</sup> /g)
Blackthorn	123.78
Hawthorn	121.42
Hazel	90.34

Six young, fully expanded leaves, without serious herbivore or pathogen damage, were selected from each hedge type (Garnier et al., 2001). Leaf samples were collected in the early morning and placed directly into plastic zip-lock bags containing damp tissue to ensure leaves remained hydrated (Garnier et al., 2001). The leaves were then digitally scanned once arranged on a white A4 sheet of card (Figure 10) around a reference of known size (a yellow dot) which was then used as a scale during digital image analysis. Leaf samples were then oven-dried at 60 degrees until constant mass was achieved (Garnier et al., 2001).



**Figure 10.** Example of scanned image of leaf sample from the hazel hedge with reference of known area (yellow dot) used to calculate the specific leaf area with digital image analysis software.

### **Coppice rotation:**

Under the managed scenarios hedges are coppiced at intervals of 15 years and modelled by removing all above-ground biomass from the system while root biomass is assumed to remain in the soil. In their model under the SRC system, Grogan and Matthews simulate plantation re-establishment by removing both above- and below-ground biomass every 24 years. In the adapted model neither managed or unmanaged hedges undergo plantation re-establishment within the simulation period (100 years). Plantation re-establishment was therefore set to occur after the period of interest within the simulations (+100 years).



### 3.3 Carbon budget analyses

Several studies indicate the importance of including carbon budgets when assessing potential management options to increase the carbon sequestration potential of forestry (Pussinen et al., 1997, 2002; Deckmyn et al., 2004). Although the reviewed literature considers the potential carbon stocks and sequestration rates of hedgerows, few studies account for the emissions from management activities and substitution of fossil fuels when hedges are managed for woodfuel. Simple carbon savings budgets which capture the emissions from management activities and the substitution of fossil fuel were therefore included in the study.

Assuming 5.33 kWh per kg of coppiced hedge material (based on woodchip calorific content analysis carried out at Elm Farm on woodchip produced from a mixed blackthorn and hazel hedge (Chambers et al, 2015)) the length of each hedge type required to produce 20,000 kWh, the typical annual energy consumption of a house (Biomass Energy Centre, 2014), was calculated.

For the managed scenarios, the potential carbon sequestration values provided by these hedge lengths over a 15 year period were calculated based on the adapted model's results. The estimated carbon emissions resulting from woodchip production were then subtracted assuming emissions of 0.14 tonnes of carbon per 20,000 kWh worth of woodchip (Biomass Energy Centre, 2014). Carbon emissions resulting from woodchip production would however, in practice, vary with hedge type, production practices, and transport distances.

As with SRC, the biomass energy from coppicing hedges is considered a carbon-neutral source of energy that does not contribute to carbon dioxide enrichment of the atmosphere (Njakou Djomo et al., 2013; Repo et al., 2011; Mann et al., 1997). The carbon emissions produced when the woodchip is burnt were therefore assumed to be zero.

For the unmanaged scenarios the potential carbon sequestration values provided by these hedge lengths over a 15 year period were once again calculated based on the model results, and the estimated carbon emissions resulting from the use of heating oil (6.28 tonnes) to provide 20,000 kWh were subtracted (Biomass Energy Centre, 2014).

## 4. Results

Here the estimated carbon stocks and flows of each hedge treatment and the results from the quantification of SOC and leaf litter inputs are presented, followed by the results from the adapted model and carbon budget analysis.

### 4.1 Estimated carbon stores and flows

Carbon stores and flows were estimated for both cut and uncut hedge plots on Elm Farm and are displayed in Table 5, 6 and 7. Due to the absence of replicates, no statistical analyses were carried out on the data; general observations only are therefore presented here as an estimate of the carbon stored under each treatment.

Carbon stores are expressed as both  $t\ C\ ha^{-1}$  and  $t\ C\ km^{-1}$  to facilitate comparison with the estimates presented in the reviewed literature. Estimates are based on a sampling depth of 50cm and the relevant hedge widths (Table 4). Estimates made on a hectare basis assume a full hectare of hedge and are not based on a set hedgerow density within the landscape.

As shown in Table 5, 6 and 7, the unmanaged hedges were estimated to store more carbon than the recently coppiced hedges. This is primarily due to the above-ground biomass having been removed, substantially decreasing above-ground carbon stores. In comparison with the hazel hedge, blackthorn and hawthorn hedges responded poorly to coppicing as shown by their lower above-ground carbon stores 1 year after coppicing.

For both hazel and hawthorn, coppiced hedges had lower leaf litter carbon flows than uncoppiced hedges. Carbon flows within the coppiced blackthorn hedge were however higher than the uncoppiced hedge. This is most likely due to the large amounts of dead twiggy material, left behind following coppicing, having been included in the leaf litter samples.

Average SOC stores were found to be higher within the unmanaged blackthorn and hawthorn hedges, if only marginally, and average SOC stores slightly higher in the coppiced hazel hedge compared to the unmanaged hazel hedge.

**Table 4. Estimated average width of hedges before coppicing and one year following coppicing.**

Estimated average hedge width (m)		
Hedge	Before coppicing	1 year after coppicing
Blackthorn	3.5	0.55
Hawthorn	6	0.70
Hazel	4	1.50

**Table 5. Estimated carbon stores and flows within and out of the system for blackthorn under both managed and unmanaged scenarios.**

	Uncoppiced hedge	1 year after coppicing	Uncoppiced hedge	1 year after coppicing
<b>Carbon stocks</b>	<b>t C ha<sup>-1</sup></b>	<b>t C ha<sup>-1</sup></b>	<b>t C km<sup>-1</sup></b>	<b>t C km<sup>-1</sup></b>
Above-ground	131.50	27.62	46.02	0.74
Below-ground	43.83	39.45	15.34	13.81
SOC	111.93	95.31	89.55	76.25
<b>Total stocks</b>	<b>287.26</b>	<b>162.38</b>	<b>150.91</b>	<b>90.80</b>
<b>Carbon flows within</b>	<b>t C ha<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C ha<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C km<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C km<sup>-1</sup> yr<sup>-1</sup></b>
Leaf litter	35.04	42.62	1.23	1.49
<b>Total flows within</b>	<b>35.04</b>	<b>42.62</b>	<b>1.23</b>	<b>1.49</b>
<b>Carbon flows out</b>	<b>t C ha<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C ha<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C km<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C km<sup>-1</sup> yr<sup>-1</sup></b>
Woodchip	0	131.50	0	46.02
<b>Total flows out</b>	<b>0</b>	<b>131.50</b>	<b>0</b>	<b>46.02</b>

**Table 6. Estimated carbon stores and flows within and out of the system for hawthorn under both managed and unmanaged scenarios.**

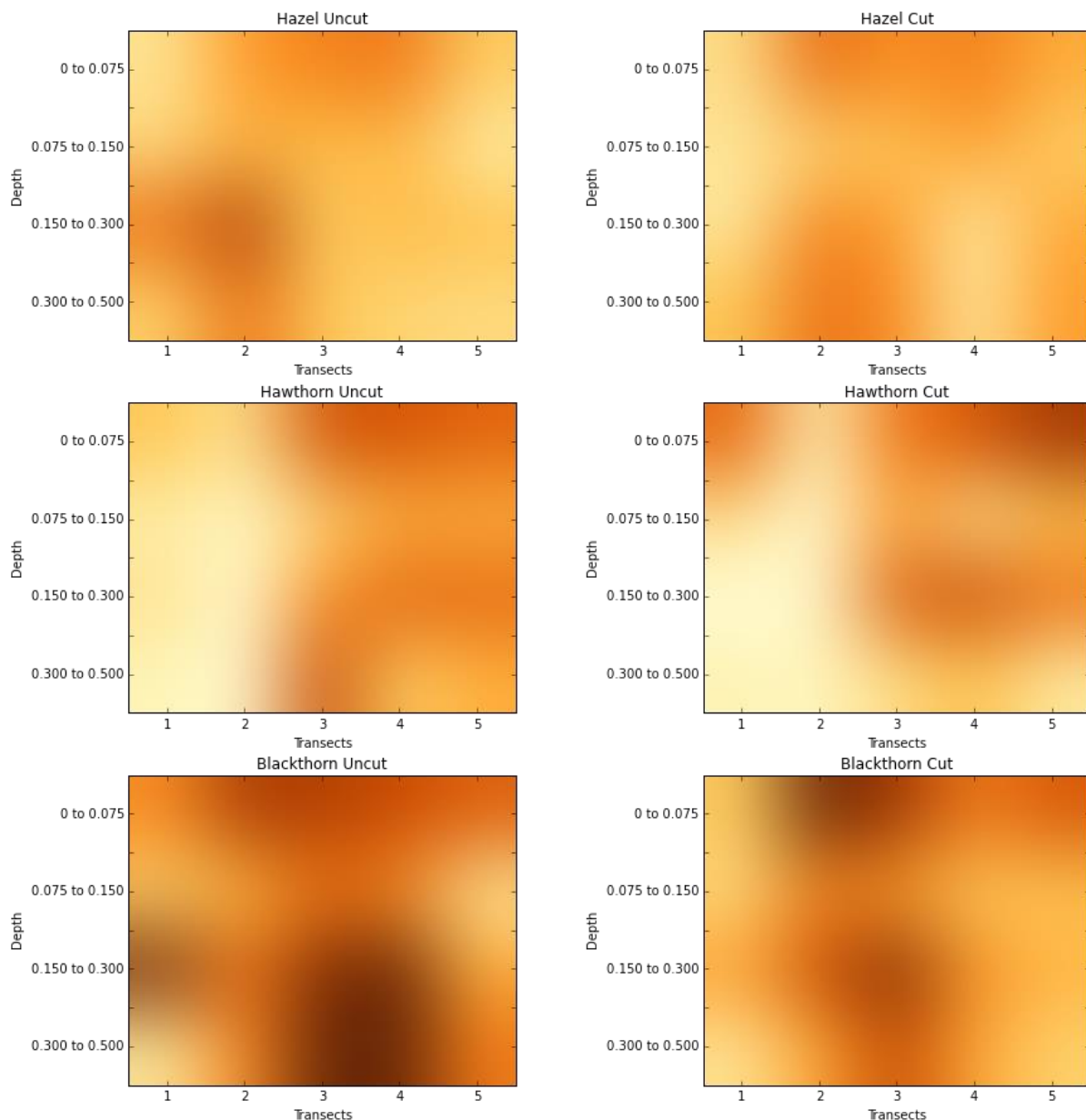
	Uncoppiced hedge	1 year after coppicing	Uncoppiced hedge	1 year after coppicing
<b>Carbon stocks</b>	<b>t C ha<sup>-1</sup></b>	<b>t C ha<sup>-1</sup></b>	<b>t C km<sup>-1</sup></b>	<b>t C km<sup>-1</sup></b>
Above-ground	93.50	25.65	28.05	0.88
Below-ground	31.17	28.05	9.35	8.42
SOC	74.04	66.52	59.23	53.22
<b>Total stocks</b>	<b>198.71</b>	<b>120.22</b>	<b>96.63</b>	<b>62.51</b>
<b>Carbon flows within</b>	<b>t C ha<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C ha<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C km<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C km<sup>-1</sup> yr<sup>-1</sup></b>
Leaf litter	32.30	26.61	0.97	0.80
<b>Total flows within</b>	<b>32.30</b>	<b>26.61</b>	<b>0.97</b>	<b>0.80</b>
<b>Carbon flows out</b>	<b>t C ha<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C ha<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C km<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C km<sup>-1</sup> yr<sup>-1</sup></b>
Woodchip	0	93.50	0	28.05
<b>Total flows out</b>	<b>0</b>	<b>93.50</b>	<b>0</b>	<b>28.05</b>

**Table 7. Estimated carbon stores and flows within and out of the system for hazel under both managed and unmanaged scenarios.**

	Uncoppiced hedge	1 year after coppicing	Uncoppiced hedge	1 year after coppicing
<b>Carbon stocks</b>	<b>t C ha<sup>-1</sup></b>	<b>t C ha<sup>-1</sup></b>	<b>t C km<sup>-1</sup></b>	<b>t C km<sup>-1</sup></b>
Above-ground	45.08	34.35	18.03	2.52
Below-ground	15.03	13.52	6.01	5.41
SOC	85.36	88.80	68.29	71.04
<b>Total stocks</b>	<b>145.46</b>	<b>136.67</b>	<b>92.33</b>	<b>78.97</b>
<b>Carbon flows within</b>	<b>t C ha<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C ha<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C km<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C km<sup>-1</sup> yr<sup>-1</sup></b>
Leaf litter	20.85	8.98	0.83	0.36
<b>Total flows within</b>	<b>20.85</b>	<b>8.98</b>	<b>0.83</b>	<b>0.36</b>
<b>Carbon flows out</b>	<b>t C ha<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C ha<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C km<sup>-1</sup> yr<sup>-1</sup></b>	<b>t C km<sup>-1</sup> yr<sup>-1</sup></b>
Woodchip	0	45.08	0	18.03
<b>Total flows out</b>	<b>0</b>	<b>45.08</b>	<b>0</b>	<b>18.03</b>

### 4.1.1 Distribution of soil organic carbon

Figure 11 shows the distribution of SOC within each hedge plot. Both cut and uncut plots of each hedge type have a similar distribution of SOC across transects and soil depths. This may suggest that the distribution of SOC is largely determined by factors other than whether or not a hedge is coppiced such as topography and soil type. However, based on the slow response and long residence times of SOC (Howlett et al., 2011; Coleman et al., 1997; Jenkinson et al., 1997; Upson and Burgess, 2013), any dramatic changes in SOC concentration and distribution are unlikely to have occurred within one year following coppicing.



**Figure 11.** Distribution of SOC within each hedge plot displayed in soil depth by transect matrices with interpolation. Darker colours represent high concentrations of SOC.

### 4.1.2 Distribution of leaf litter inputs

For each hedge type, leaf litter distribution following coppicing would seem to have become narrower and more concentrated within transects close to the coppice stools (Figure 12). This is as expected due to uncoppiced hedges having a wider area of leaf fall due to their larger canopies, while the hedge canopy is severely reduced following coppicing.

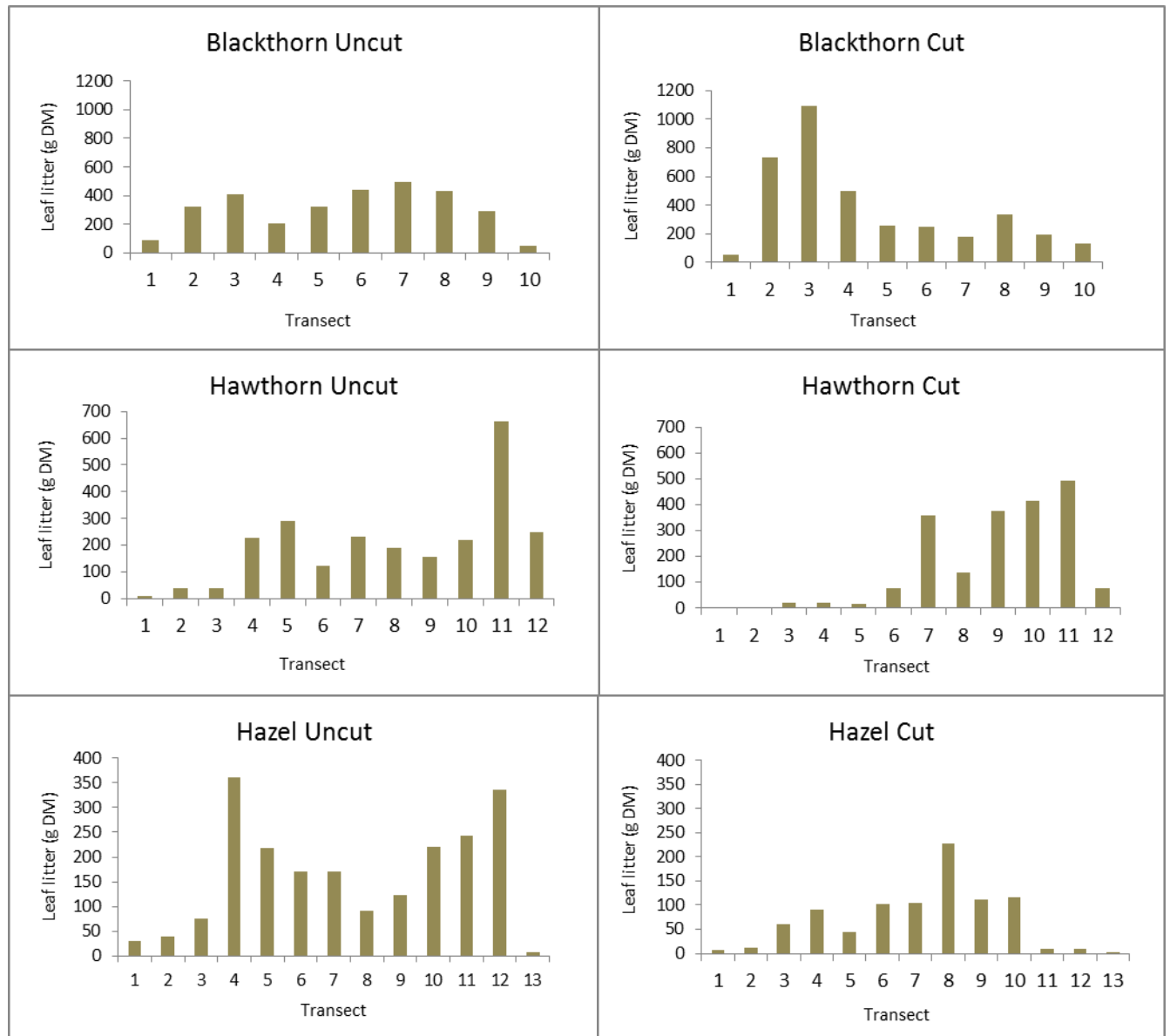


Figure 12. Distribution of leaf litter inputs within each hedge plot.

## 4.2 Modelling results

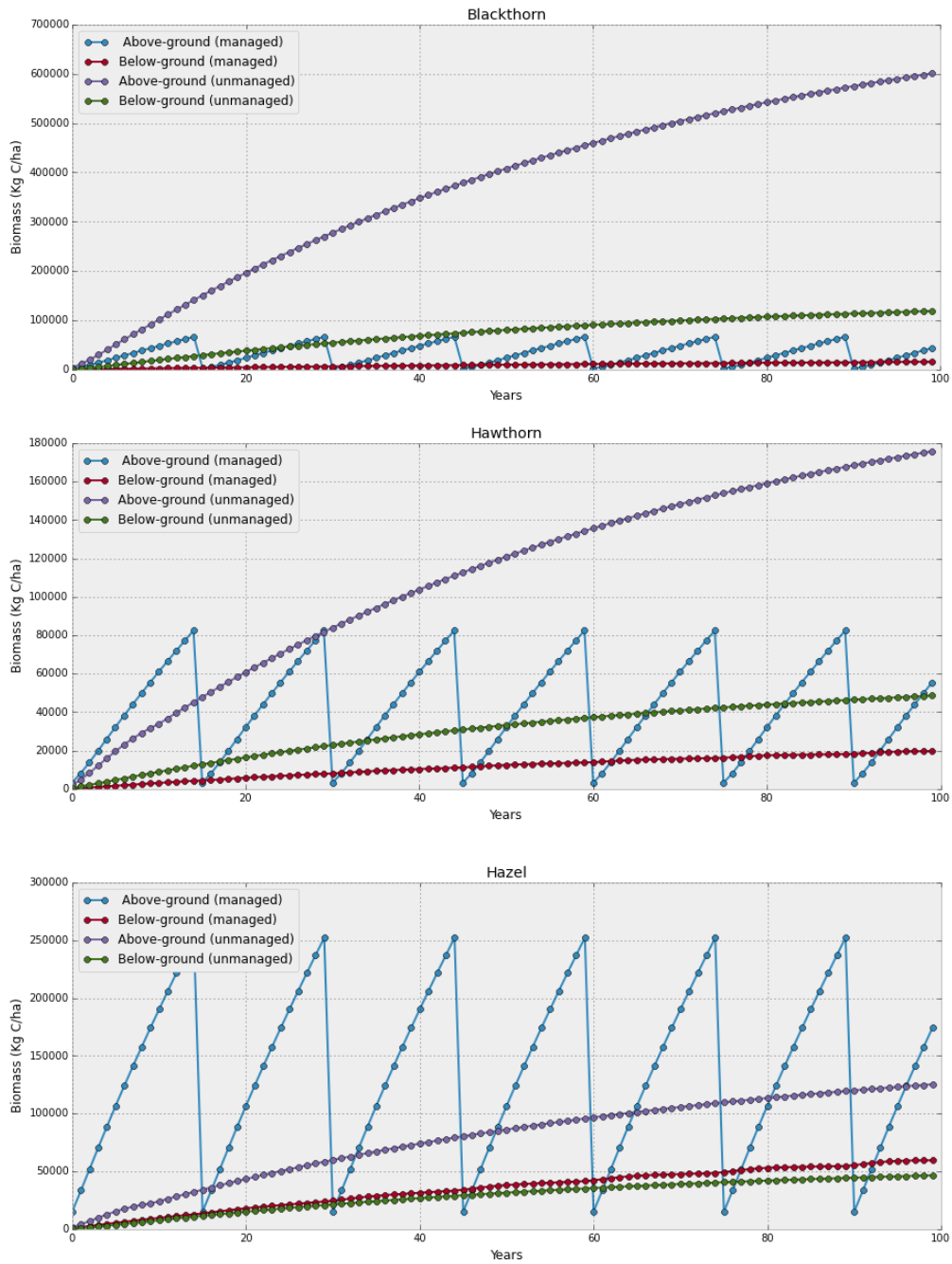
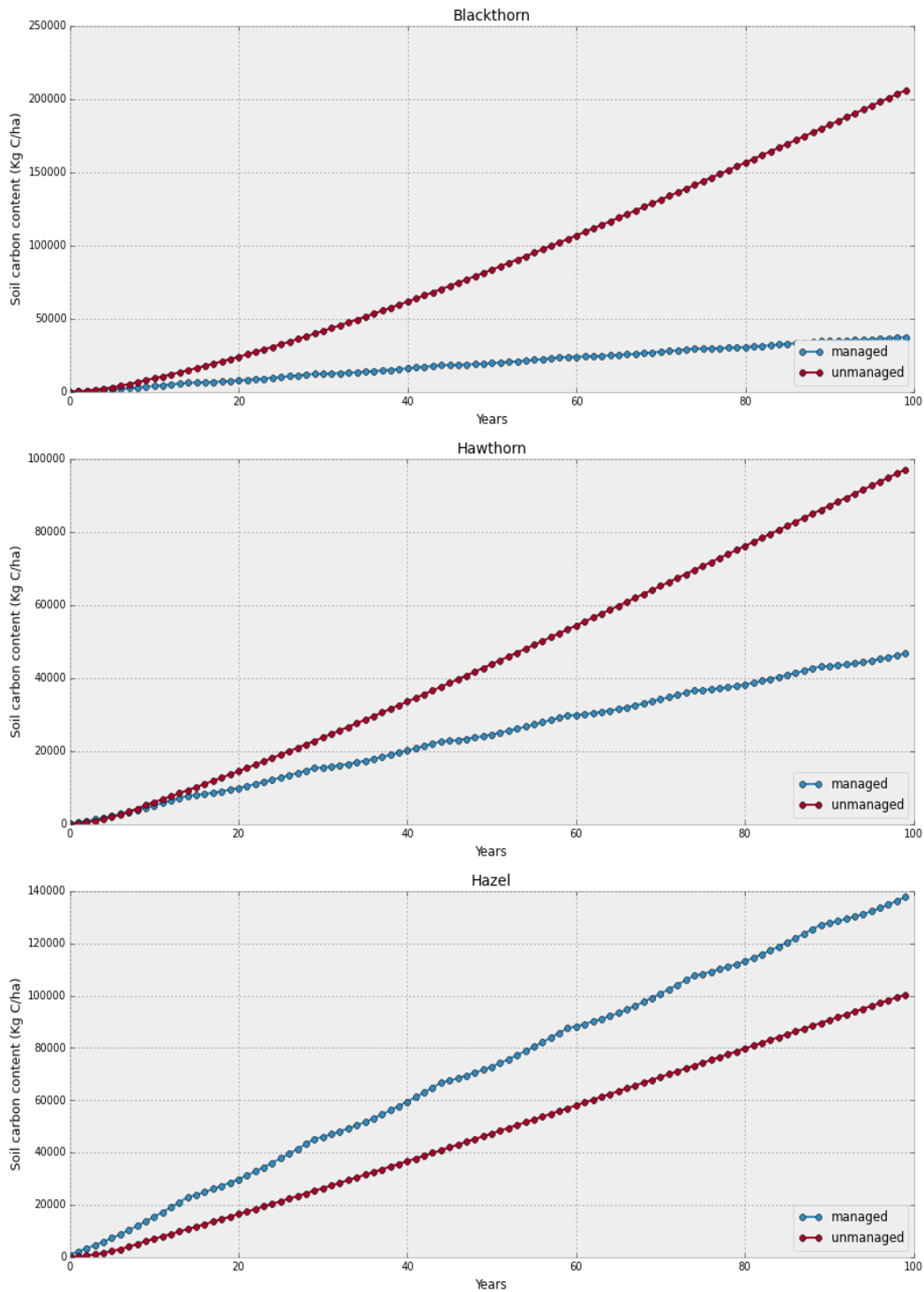


Figure 13. Predicted above- and below-ground biomasses for both managed and unmanaged scenarios for each hedge type over 100 years. Managed hedges are coppiced in 15-year intervals.



**Figure 14. Predicted soil carbon for both managed and unmanaged scenarios for each hedge type over 100 years. Managed hedges are coppiced in 15-year intervals.**

For blackthorn and hawthorn simulations, unmanaged scenarios were shown to sequester more carbon both in above- and below-ground biomass and SOC than managed scenarios (Figure 13 and 14). This is due to their poorer responses to coppicing when compared to the hazel hedge. The managed hazel scenario however was shown to sequester larger amounts of carbon in both below-



ground biomass and SOC over the 100 year simulation due to its exceptionally good response to coppicing (Figure 13 and 14). It is however important to note that the managed scenarios were based on production data from just one year after coppicing and assume a linear increase in biomass. In practice this may not be the case, and blackthorn and hawthorn species may just be slow to respond to coppicing. Only with continued long term monitoring and inclusion of further data points can the accuracy of the model be improved.

#### 4.2.1 Average annual carbon sequestration

Annual carbon sequestration rates were determined using a one-step linear interpolation of the graphs presented in Figure 13 and 14. The sequestration rates shown in Table 8, 9 and 10 are therefore approximated average annual carbon sequestration rates over 100 years. Carbon sequestration within above-ground biomass under managed scenarios are however assumed to be zero as above-ground biomass is removed and burnt every 15 years and therefore does not store carbon in the long-term. Values are expressed in both  $\text{t C ha}^{-1}$  and  $\text{t C km}^{-1}$ . As with the previous carbon storage and flow estimates, values on a hectare basis assume a full hectare of hedge and are not based on a set hedgerow density within the landscape.

**Table 8. Approximate average annual carbon sequestration rates (displayed in both  $\text{t C ha}^{-1} \text{ yr}^{-1}$  and  $\text{t C km}^{-1} \text{ yr}^{-1}$ ) over 100 years for both managed and unmanaged blackthorn scenarios. Carbon sequestration within above-ground biomass under the managed scenario is assumed to be 0 as above-ground biomass is removed every 15 years.**

	Unmanaged $\text{t C ha}^{-1} \text{ yr}^{-1}$	Managed $\text{t C ha}^{-1} \text{ yr}^{-1}$	Unmanaged $\text{t C km}^{-1} \text{ yr}^{-1}$	Managed $\text{t C km}^{-1} \text{ yr}^{-1}$
Above-ground biomass	6.00	0.00	2.10	0.00
Below-ground biomass	1.10	0.15	0.39	0.01
SOC	2.10	0.38	0.74	0.02
Total	9.20	0.53	3.22	0.03

**Table 9. Approximate average annual carbon sequestration rates over 100 years (displayed in both  $\text{t C ha}^{-1} \text{ yr}^{-1}$  and  $\text{t C km}^{-1} \text{ yr}^{-1}$ ) for both managed and unmanaged hawthorn scenarios. Carbon sequestration within above-ground biomass under the managed scenario is assumed to be 0 as above-ground biomass is removed every 15 years.**

	Unmanaged $\text{t C ha}^{-1} \text{ yr}^{-1}$	Managed $\text{t C ha}^{-1} \text{ yr}^{-1}$	Unmanaged $\text{t C km}^{-1} \text{ yr}^{-1}$	Managed $\text{t C km}^{-1} \text{ yr}^{-1}$
Above-ground biomass	1.79	0.00	0.54	0.00
Below-ground biomass	0.50	0.20	0.15	0.01
SOC	9.90	0.43	2.97	0.03
Total	12.19	0.63	3.66	0.04

**Table 10. Approximate average annual carbon sequestration rates over 100 years (displayed in both  $\text{t C ha}^{-1} \text{ yr}^{-1}$  and  $\text{t C km}^{-1} \text{ yr}^{-1}$ ) for both managed and unmanaged hazel scenarios. Carbon sequestration within above-ground biomass under the managed scenario is assumed to be 0 as above-ground biomass is removed every 15 years.**

	Unmanaged $\text{t C ha}^{-1} \text{ yr}^{-1}$	Managed $\text{t C ha}^{-1} \text{ yr}^{-1}$	Unmanaged $\text{t C km}^{-1} \text{ yr}^{-1}$	Managed $\text{t C km}^{-1} \text{ yr}^{-1}$
Above-ground biomass	1.25	0.00	0.50	0.00
Below-ground biomass	0.49	0.65	0.20	0.10
SOC	1.0	1.39	0.40	0.21
<b>Total</b>	<b>2.74</b>	<b>2.04</b>	<b>1.10</b>	<b>0.31</b>

### 4.3 Carbon budget analyses

Assuming 5.33 kWh per kg of coppiced hedge material (based on calorific content analysis carried out at Elm Farm on woodchip produced from a mixed blackthorn and hazel hedge) the length of each hedge type required to produce 20,000 kWh, the typical annual energy consumption of a house (Biomass Energy Centre, 2014), was calculated (Table 11).

**Table 11. Length of each hedge type required to produce 20,000 kWh, the typical annual energy consumption of a house (Biomass Energy Centre, 2014).**

Hedge type	Metres required annually
Blackthorn	40
Hawthorn	66
Hazel	102

To calculate the carbon budgets for the managed scenarios (Table 12, 13 and 14) the potential carbon sequestration values provided by these hedge lengths over a 15 year period were then calculated based on the model results and the estimated carbon emissions resulting from woodchip production subtracted. Carbon emissions for woodchip production were assumed to be 0.14 tonnes of carbon per 20,000 kWh worth of woodchip based on figures presented by the Biomass Energy Centre (2014). Carbon emissions resulting from woodchip production would however, in practice, vary with hedge length, type, production practices, and transport distances.

As with SRC, the biomass energy from coppicing hedges is considered a carbon-neutral source of energy that doesn't contribute to carbon dioxide enrichment of the atmosphere (Njakou Djomo et al., 2013; Repo et al., 2011; Mann et al. 1999). The carbon emissions produced when the woodchip is burnt were therefore assumed to be zero.

For the unmanaged scenario carbon budgets, the potential carbon sequestration values provided by these hedge lengths over a 15 year period were once again calculated based on the model results, and the estimated carbon emissions resulting from use of heating oil (fossil fuel) subtracted. Carbon emissions resulting from use of heating oil were assumed to be 6.28 tonnes of carbon per year to heat a typical house (Biomass Energy Centre, 2014).

Despite all unmanaged hedges sequestering more carbon, all three hedge types save more carbon when managed for woodfuel than when left unmanaged due to the substitution of fossil fuels, as shown by the simple carbon budget results in Table 12, 13 and 14.

**Table 12. Simple carbon budget for both managed and unmanaged blackthorn scenarios and the potential carbon savings when managed for woodfuel.**

	Managed (t C yr <sup>-1</sup> )	Unmanaged (t C yr <sup>-1</sup> )
C sequestered	0.02	1.93
C released	0.14	6.28
Total carbon sequestered	0.12	-4.35
Carbon saving when hedge managed for woodfuel: 4.47 t C yr <sup>-1</sup>		

**Table 13. Simple carbon budget for both managed and unmanaged hawthorn scenarios and the potential carbon savings when managed for woodfuel.**

	Managed (t C yr <sup>-1</sup> )	Unmanaged (t C yr <sup>-1</sup> )
C sequestered	0.04	3.62
C released	0.14	6.28
Total carbon sequestered	-0.10	-2.66
Carbon saving when hedge managed for woodfuel: 2.56 t C yr <sup>-1</sup>		

**Table 14. Simple carbon budget for both managed and unmanaged hazel scenarios and the potential carbon savings when managed for woodfuel.**

	Managed (t C yr <sup>-1</sup> )	Unmanaged (t C yr <sup>-1</sup> )
C sequestered	0.47	1.68
C released	0.14	6.28
Total carbon sequestered	0.33	-4.60
Carbon saving when hedge managed for woodfuel: 4.93 t C yr <sup>-1</sup>		

## 5. Discussion

The discussion is structured around the three methods used in determining the effects of managing hedges for woodfuel on carbon sequestration: estimates of carbon stores and flows, model adaptation, and carbon budgets, and is followed by a conclusion based on the report's findings and supporting literature on the carbon sequestration potential of UK hedges managed for woodfuel.

### 5.1 Estimation of current carbon stores and flows

This study indicates that carbon stores within unmanaged hedges, that is, hedges not managed by coppicing, are higher than within hedges one year following coppicing, with most of this difference due to the substantial decrease in above-ground biomass following coppicing. Carbon stored within the above-ground biomass was estimated to be higher in the unmanaged hedges by an average of  $60.82 \text{ t C ha}^{-1}$  and  $29.32 \text{ t C km}^{-1}$  when compared with hedges one year after coppicing. This large difference in estimated above-ground carbon stores would however become smaller as the recently coppiced hedges re-grow, and continue sequestering carbon within their above-ground biomass.

Above-ground carbon stores within coppice regrowth were based on the production data from unmanaged hedges when harvested. This method assumed coppice regrowth to be of the same density as material harvested from the mature unmanaged hedges, which in practice is very unlikely. These above-ground carbon estimates are therefore likely to have been overestimated. To acquire more accurate data on the productivity of hedges following coppicing chronological studies using destructive methods or non-destructive estimation methods, such as hedgerow-specific allometric equations, would be needed.

As expected, based on the reduction of above-ground biomass, both coppiced hawthorn and hazel hedges were found to have lower leaf litter carbon flows than uncoppiced hedges. Carbon flows within the coppiced blackthorn hedge were however higher than the uncoppiced blackthorn hedge. This is most likely due to the large amounts of dead twiggy material left behind following coppicing, having been included in the leaf litter samples.

Average SOC stores were found to be higher within the uncoppiced blackthorn and hawthorn hedges, if only marginally, and average SOC stores slightly higher in the coppiced hazel hedge compared to the uncoppiced hazel hedge. Due to the absence of replicates, no statistical analyses were carried out on the data; general observations only are therefore possible. Although differences in SOC between coppiced and uncoppiced hedge plots are observed, these differences would seem negligible with an average difference of  $7.35 \text{ t C ha}^{-1}$  equivalent to  $9.19 \text{ t C km}^{-1}$ . Similarly, little variation in SOC distribution was seen between coppiced and uncoppiced plots for each hedge species. These limited differences in SOC concentration and distribution may be explained by the slow response and long residence times of SOC (Howlett et al., 2011; Coleman et al., 1997; Jenkinson et al., 1997; Upton and Burgess, 2013). Dramatic changes in SOC are therefore unlikely to have occurred within one year of coppicing. Once again, chronological studies would be required to establish how SOC changes between coppice intervals and to observe any long-term impacts of coppicing on carbon storage.

Current models reviewed within the literature estimate the above-ground biomass carbon stocks of typical agricultural hedges, synonymous to unmanaged hedges within this study, to range from  $5 \text{ t C ha}^{-1}$  to  $45 \text{ t C ha}^{-1}$  (Falloon et al., 2004; Warner, 2011; Robertson et al., 2012) and SOC stores from  $43 \text{ t C ha}^{-1}$  to  $136.8 \text{ t C ha}^{-1}$  (Falloon et al., 2004; Robertson et al., 2012). With an average of  $90.02 \text{ t C ha}^{-1}$ , this study's estimated above-ground biomass carbon stores are considerably higher than previously published estimates. This may be explained, not only by the variable nature of hedges, but by the particularly tall overgrown character of the hedges at Elm Farm due to relaxed management over the past decade. Soil organic carbon stores presented by this study for

unmanaged hedges, however, range from 74.04 t C ha<sup>-1</sup> to 111.93 t C ha<sup>-1</sup>, well within the ranges previously reported.

Within this study several flows and stores of carbon were not included due to difficulties with their measurement and estimation, for example soil respiration and carbon stored within the unutilised biomass of the coppice stools. Due to limited data on root biomass production and dynamics for the hedge species investigated, root biomass was simply assumed to be a third of the estimated above-ground biomass and to decrease by 10% following coppicing due to necrosis. In addition, hedgerow trees are a common feature of European hedgerows (Forman and Baudry, 1984; Auclair and Dupraz, 1999; Barr, 2004; Ryszkowski and Kedziora, 2007) and, with large volumes of above-ground biomass, are likely to contribute significantly to the potential carbon storage of hedgerows (Wolton et al., 2014). Hedgerow trees were however not included within this study. Such assumptions and unaccounted flows and stores are therefore likely to have resulted in inaccuracies within the resulting estimated carbon stores and flows. The presented estimates of carbon stores and flows from this small-scale, short-term study of hedges managed for woodfuel should therefore be viewed with caution, although they may serve as a useful starting point for future investigation.

## 5.2 Model adaptations

For all three hedge types, unmanaged scenarios were shown to have higher carbon sequestration rates than the managed scenarios by an average of 6.98 t C ha<sup>-1</sup> yr<sup>-1</sup> and 2.53 t C km<sup>-1</sup> yr<sup>-1</sup>. These differences are largely attributed to the larger above-ground carbon stores of the unmanaged hedges based on the assumption that the above-ground biomass of the managed hedges does not contribute to the long-term storage of carbon when burnt for energy production. Above-ground biomass harvested from hedges under the management scenario would, however, contribute to carbon savings through the substitution of fossil fuels when used for bioenergy and is taken into account by the carbon budget analyses.

Carbon sequestration rates modelled by Taylor et al. (2010) for non-flailed hedges range from 2.20 t C ha<sup>-1</sup> yr<sup>-1</sup> to 11.40 t C ha<sup>-1</sup> yr<sup>-1</sup> with a mid-range of 6.37 t C ha<sup>-1</sup> yr<sup>-1</sup>. Although substantially higher than those presented by Falloon et al. (2004) (shrubby hedge = 1 t C ha yr<sup>-1</sup>, lines of trees = 2.8 t C ha yr<sup>-1</sup>), it is encouraging to note that the carbon sequestration rates modelled from this study for unmanaged hedge scenarios, ranging from 2.74 t C ha yr<sup>-1</sup> to 12.19 t C ha yr<sup>-1</sup>, are in line with those made by Taylor et al. (2010).

Leaf area index (LAI) values for both managed and unmanaged hedge scenarios were taken from Pocock et al. (2010) who developed a predictive model for hedgerow LAI based on measured data. Under both managed and unmanaged scenarios LAI was assumed to increase linearly up to a maximum of 6.8 m<sup>2</sup>/m<sup>2</sup>. LAI are however likely to vary with hedge species and management practices and unlikely to increase linearly. Additionally, annual increases in LAI values for managed scenarios were based on the first year's regrowth data of coppiced hedge plots. The higher LAI values for the hazel and hawthorn managed scenarios therefore reflect the faster growth rate due to the invigorating effects of coppicing (Dickman et al., 1996). For blackthorn, however, regrowth one year following coppicing was poor. This may suggest blackthorn is a less suitable species for coppice management or it may be the case that blackthorn is simply slower to respond to coppicing and growth rates may pick up a couple of years following cut back. Further investigation into the species specific responses to coppicing and resulting LAI is therefore needed.

A substantial flaw in the adapted model is that both managed and unmanaged scenarios are simulated from hedge establishment despite using data from mature (approx. 20 to 30 years old) hedges which were then coppiced. This is due to Grogan and Matthews' original model having been developed to compare SRC from establishment and the naturally regeneration of woodland. In practice, both managed and unmanaged hedges would have similar initial growth rates after

planting and before the first coppice rotation. In future adaptations of the model this would need to be addressed for fairer comparison of managed and unmanaged scenarios.

In summary, the adapted hedgerow model is still heavily dependent on assumptions, and the results presented by this study should therefore be seen as a preliminary and used only to guide further enquiry. Adaptation of Grogan and Matthews' model to a hedgerow scenario has, however, identified where additional data collection is required for the improvement of carbon sequestration models for hedgerow systems.

### 5.3 Carbon budgets

Despite all unmanaged hedge scenarios sequestering more carbon than managed scenarios, results from the carbon budget analyses revealed that all three managed scenarios save more carbon than unmanaged scenarios due to the substitution of fossil fuels via the production of woodfuel. Considering all three unmanaged hedge scenarios were shown by the adapted model to have higher carbon sequestration rates than the managed scenarios, these results highlight the importance of including carbon budgets and accounting for carbon substitution when assessing the impacts of potential management options for climate change mitigation.

The carbon budget analyses within this study however rely on a number of generalisations and assume similar carbon emissions as those reported for other woodchip production systems. To improve the reliability of these carbon budgets, not only is further empirical data on carbon dynamics under hedgerow systems needed, but further quantification of the embedded energy in the production of woodfuel from such systems is required.

## 6. Conclusion

The validity of current estimates for hedgerow carbon stocks and accumulation rates is limited by incomplete information on the effect of vegetation type, hedge structure, management practices, carbon budgets, and the landscape-scale impact of carbon storage processes. This study aimed to address a number of these shortfalls and to explore the effects of hedgerow management for woodfuel on carbon sequestration.

The study revealed that while hedges which are not managed by coppicing sequester larger quantities of carbon, total carbon savings are higher when hedges are managed by coppicing due to the substitution of fossil fuels via the production of woodfuel, and highlights to importance of whole system carbon budgets.

Although the results presented by this small-scale, short-term study should be viewed as provisional, they present a useful starting point for future enquiry, identifying the need for long-term chronological studies and data collection on carbon sequestration processes specific to hedges. It is also encouraging to note that estimates for carbon sequestration under unmanaged scenarios presented here are in line with those found in other studies. Collection of further empirical data on the carbon sequestration potential of hedgerows will however be needed to validate existing estimates and models and to inform decisions not only at a farm management level but also for wider policy.

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