

**Review of the potential for soil carbon  
sequestration under bioenergy crops in the  
U.K.**

**Scientific Report**

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The establishment of bioenergy crop plantations of short rotation coppice trees and miscanthus is intended as one component of a government strategy to mitigate net CO<sub>2</sub> emissions in accordance with U.K. commitments to the U.N. Kyoto Protocol on Climate Change. In addition to the provision of "carbon neutral" fossil fuel replacement "energy crops", bio-energy plantations may also contribute to CO<sub>2</sub> emissions mitigation by enhancing sequestration of carbon in soil organic matter in the underlying soils.

First, we review available literature and on-going research, both in the U.K. and internationally, in order to evaluate current knowledge on the potential for soil carbon sequestration under bioenergy crops. Our analysis indicates that conclusive experimental data on soil carbon sequestration is currently constrained by the short time since most of the plantations have been established. At present, the experimental data indicate that soil carbon sequestration rates under short rotation coppice plantations can range from 0-1.6 Mg C ha<sup>-1</sup> y<sup>-1</sup>.

Second, we review the potential for established soil carbon models to be adapted for use in a bioenergy crop system context to predict soil carbon sequestration rates. Of all available models, CENTURY seemed to have the best potential for adaptation to bioenergy crop systems because of its integrated plant-soil approach, and the availability of specific forestry sub-routines for the model.

Third, we develop a simple model using a carbon mass balance approach to predict soil carbon sequestration. The model is site-specific and calibrated to soil carbon data from a natural woodland regeneration site in the U.K. According to the model output, we conclude that there is potential for significant soil carbon sequestration in short-rotation coppice plantations in the U.K. In this very preliminary and site-specific analysis that necessitated several major assumptions, soil carbon sequestration rate under willow was estimated as 0.5 Mg C ha<sup>-1</sup> y<sup>-1</sup>. The model identifies the following factors as being major controls on rates and amounts of soil carbon sequestration under coppiced willow: carbon inputs (net primary production); decomposition rates of the major soil carbon pools, initial soil carbon content (an inverse relationship with soil carbon sequestration); crop/plantation management practice; and depth of soil being influenced by the bioenergy crop. Carbon sequestration is most likely on soils whose carbon content has been depleted to relatively low levels due to previous management practices. Carbon sequestration in soil is likely to occur until the original climatically-controlled equilibrium point between soil carbon inputs and outputs is reached.

Considerable further research would be required to develop a general model that could incorporate climatic and soil-type variation, as well as hydrological interactions associated with the relatively high water demand of bioenergy crops, to accurately predict the potential for soil carbon sequestration in bioenergy plantations across the U.K.

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# 1 Introduction

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## 1.1 Bioenergy crops for mitigation of CO<sub>2</sub> emissions

The United Nations Framework Convention on Climate Change Kyoto Protocol of 1997 was a major landmark in international strategy to combat rising atmospheric CO<sub>2</sub> concentrations. Following the Protocol, the U.K. committed itself to an 8% reduction in CO<sub>2</sub> emissions from the 1990 baseline level during the first commitment period (2008-2012) (DETR 1998). Considerable effort has since been placed into estimating national carbon source and sink fluxes for all major land-use practices in the U.K., as well as the effects of changes in land-use on those fluxes (Cannell, Milne et al. 1999). In 1990, biological activities in U.K. terrestrial ecosystems resulted in a net release of approximately 4Tg C y<sup>-1</sup> (T = 10<sup>12</sup>) to the atmosphere; small in comparison to the 159 Tg C emitted by the anthropogenic burning of fossil fuels in the same year (Cannell, Milne et al. 1999). Thus, the 8% target agreed by the U.K. in Kyoto indicates that that this country is committed to an average annual reduction of 12.7 Tg C in national emissions during 2008-2012. Clearly, the most effective short-term means of reducing emissions to reach this target would be to reduce fossil fuel use by encouraging better energy use efficiency. In addition, conversion to renewable bioenergy fuel sources, and the adoption of techniques to capture CO<sub>2</sub> as it is being released from power stations are also being tested as appropriate measures to reduce net emissions. As an alternative approach, the Protocol encourages demonstrable sequestration of atmospheric carbon through changes in land use/management as a means to offset national CO<sub>2</sub> emissions. Soils and litter represent the largest carbon pools in the U.K. (estimated at 10,000 Tg C, half of which is in the form of peat mainly in Scotland) (DETR 2000). Relatively slow rates of carbon turnover within the principal soil organic pools (Paul and Clark 1996) imply that even small increases in total carbon stored in soil will have a particularly long-term mitigation effect on net carbon emissions. Therefore, in addition to reducing net emissions by encouraging increased energy efficiency, and the use of non-fossil fuel energy sources, the U.K. government is strongly promoting sustainable forestry and farming as means of sequestering carbon in woody biomass and in soil organic pools (DETR 2000).

Short rotation coppice (SRC) willow plantations have been identified as the agricultural land use change with the greatest potential for carbon mitigation across the U.K. (Smith, Powlson et al. 2000). Willow is an ideal species for short rotation coppice in the U.K. because of its suitability to regional climate and soil conditions, and its vigorous shoot regeneration after coppicing. Current recommendations for SRC plantation establishment are that willow stem cuttings are planted at densities of 20,000 ha, that the stems are coppiced after one year to promote multi-stemming, and that harvesting should take place during winter every three years afterwards (Dawson 2000), (Armstrong 2000). Predicted yields range from 10-18 t DM (tonnes dry matter) ha<sup>-1</sup> y<sup>-1</sup> (~4-7.2 Mg C ha<sup>-1</sup> y<sup>-1</sup>) (Armstrong 2000), (Taylor and Bunn 2000). Since repeated short-term coppicing is likely to deplete plant capacity for shoot regeneration over time, it is expected that plantations will probably need to be completely replaced every 20-30 years (Britt 1995).

Miscanthus grass plantations have also been suggested as having potential for a substantial beneficial effect in relatively warm and dry areas of the U.K. (Rutherford and Heath 1992). Miscanthus is a C4 perennial grass originating from Asia. Two species have been used in bioenergy crop plantations: *Miscanthus X giganteus* (a sterile hybrid) and *Miscanthus sacchiflorus*. Current recommendations for miscanthus bioenergy crop plantations are that rhizomes are planted at densities of 20,000 ha<sup>-1</sup> in late spring, and that harvesting begins a year later (when the crop may have reached a height of 2-3 m) (Bullard 1999). Annual harvests continue to increase for at least 5 years at which point yields can range from 12-25 t DM ha<sup>-1</sup> y<sup>-1</sup> (~4.8-10 Mg C ha<sup>-1</sup> y<sup>-1</sup>) (Bullard 1999).

## 1.2 Soil carbon sequestration under bioenergy crops

Much of the carbon mitigation potential associated with the use of SRC willow and miscanthus as bioenergy crops arises from their indefinite capacities as 'carbon neutral' alternatives to fossil fuel combustion. In other words, energy may be generated indefinitely using bioenergy crops with little overall impact on net carbon emissions. Nevertheless, bioenergy plantations will require energy inputs associated with initial crop establishment, fertilisation, harvesting, transport and milling that are most likely to be supplied from fossil fuels. Analyses of the overall energy balance suggest that energy output to input ratio would range from 10:1 to 20:1 (Lewandowski, Clifton-Brown et al. 2000), (Cook and Beyea 2000).

In addition to being 'carbon neutral' fossil fuel alternatives, the use of bioenergy crops on previously agricultural soil could lead to finite, but substantial increases in stored carbon pools below-ground (Smith, Powlson et al. 2000). Such increases may result from larger quantities of rootstock below-ground as well as enhanced soil organic matter carbon content. Thus, the below-ground carbon pools that might be affected by bioenergy crop plantations represent a continuum from living root biomass and freshly senesced leaf and root litter material, some of which is ultimately converted to increasingly more recalcitrant (i.e. difficult to decompose) soil humic fractions (Paul and Clark 1996). The rates of carbon processing activity in each of these pools is progressively lower (Paul and Clark 1996), implying that from a carbon mitigation perspective, the greatest long-term benefit arises from sequestration in the more recalcitrant soil carbon pools. However, the mechanisms underlying any of these carbon storage processes in bioenergy plantations are little investigated, and not well understood. Indeed, it has recently been concluded that the principal uncertainties associated with bioenergy crops for carbon mitigation are those associated with soil carbon sequestration (DETR 2000), (Cannell, Milne et al. 1999). Consequently, confidence in the predictions for the overall carbon mitigation impact of bioenergy crops is severely limited. In this review of current data and modelling approaches, we consider all processes that contribute to changes in carbon content below-ground underneath

bioenergy crops, including increased root biomass (carbon accumulation) as well as changes in the major soil organic matter carbon pools (soil carbon sequestration).

### 1.3 The aims and scope of this report

This report reviews available literature and current research on soil carbon sequestration under short rotation coppice and miscanthus bioenergy crops in the U.K., and internationally. The above-ground aspects of fuel production in short rotation coppice tree and miscanthus plantations is currently being studied in the U.K. (Forestry Commission and ADAS), continental Europe (e.g. Sweden, Denmark, and Germany) and the U.S. This report reviews these experiments, as well as suggesting future research priorities. In addition, the report reviews and evaluates the potential for using mechanistic, process-based models (such as 'Century' and 'RothC') to predict soil carbon sequestration under both bioenergy crops. Ultimately, policy formulation on bioenergy crops will require a robust model that includes site-specific variables such as local climate, soil physical properties and nutrient contents, and initial soil organic carbon contents to predict below-ground carbon sequestration potential. Furthermore, since harvesting implies ongoing depletion of soil nutrients, management of bioenergy crop plantations will also require a model that runs through a series of harvest cropping cycles in order to predict when and how much fertilisation will be required. One of the aims of this review is to establish whether there is an appropriate process-based model already available which could provide a framework for predicting the carbon mitigation potential and management requirements of bioenergy crop plantations.

Furthermore, since the Kyoto Protocol specifies a series of commitment periods, policy formulation on the implementation of bioenergy crop plantations for carbon mitigation will be strongly influenced by the temporal dynamics of the relative contributions of each carbon mitigation component. In other words, a relatively fast sequestration of carbon in plant tissue above- *and* below-ground may have a particularly important initial carbon mitigation effect even though the principal long-term carbon mitigation benefits will be derived from the replacement of fossil fuel use after subsequent harvests. Such an early rapid sequestration of carbon above- and below-ground could be crucial in satisfying the U.K.'s Kyoto Protocol commitments in the early years of adoption of bioenergy fuel plantations. Accordingly, this review will assess how the relative contributions of the two carbon mitigation components ('carbon neutral' fossil fuel replacement and soil carbon sequestration) change over time following establishment of the plantations. This exercise will facilitate appropriate comparisons of the mitigation potential of each crop, and of the timing of potential biofuel and soil carbon sequestration contributions to carbon mitigation in bioenergy plantations in the U.K.

The specific objectives of this report are:

**Objective 1.** To review available literature and evaluate ongoing research relating to soil carbon sequestration under bioenergy crops.

**Objective 2.** Review and evaluate the potential for established process-based soil carbon models to predict soil carbon sequestration under bioenergy crops.

**Objective 3.** Evaluate the relative contributions from the fossil fuel replacement and soil carbon sequestration components of carbon mitigation from the time of bioenergy crop plantation establishment through successive harvests.

**Objective 4.** Identify future research priorities to determine the impact of bioenergy crop plantations on soil carbon sequestration.

## 2 Current status of research into soil C sequestration under bioenergy crops

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### 2.1 Short rotation coppice trees

#### 2.1.1 Potential for soil carbon sequestration in the U.K.

The carbon mitigation potential of a range of agricultural practices, including willow plantations on 'set-aside' arable land, has recently been estimated for the U.K. (Smith, Powlson et al. 2000) followed by a revised version (Smith, Milne et al. 2000), and for the E.U. (Smith, Powlson et al. 2000). These studies conclude that bioenergy tree coppice plantations provide the greatest potential amongst all feasible agricultural land-management strategies. The carbon mitigation potential of bioenergy crops is derived from two distinct components: i) the replacement of fossil fuel by bioenergy crops in electricity generation stations; and ii) the sequestration of soil organic carbon beneath bioenergy crops (Smith, Powlson et al. 2000). The extra advantage of bioenergy tree coppice production over all other scenarios is that the fossil fuel carbon substitution component continues indefinitely. On an annual basis, the fossil fuel substitution component represents the major contribution to carbon mitigation potential associated with bioenergy crops in the U.K. (2.80 Tg C y<sup>-1</sup>). Nevertheless, it is estimated that increased soil organic carbon content beneath bioenergy tree coppice plantations will constitute one fifth of the total carbon mitigation potential (0.66 Tg C y<sup>-1</sup>) of bioenergy crop plantations (Smith, Milne et al. 2000). These figures are calculated assuming that 10% of arable land is converted to bioenergy crop production. Therefore, the actual rate of soil carbon sequestration under bioenergy crops used in these analyses is equivalent to 0.98 Mg C ha<sup>-1</sup> y<sup>-1</sup> (Smith, Powlson et al. 2000), (Smith, Milne et al. 2000). In short, the study of Smith *et al.* concludes that soil carbon sequestration beneath bioenergy tree coppice plantations could make a substantial contribution to the overall carbon mitigation potential of future land-management practices in the U.K.

In evaluating the conclusions of Smith *et al.*'s first U.K. study (2000b) and the revised version (Smith, Milne et al. 2000), it is necessary to identify and understand the following major assumptions that have been incorporated in developing the extrapolations.

##### 2.1.1.1 Rate of soil carbon sequestration

The estimates of carbon mitigation potential under short rotation coppice tree plantations are based on extrapolation from a single research station in the U.K. (Smith, Powlson et al. 2000). Soil organic carbon data were available from a site at Geescroft Wilderness (Poulton 1996), and from a nearby site at Broadbalk Wilderness (Jenkinson 1990), that had both been allowed to undergo natural woodland regeneration since 1885 and 1880 respectively. By comparing the initial soil organic carbon contents to 30 cm depth with those after 102 and 100 years respectively, it was possible to estimate the average annual increase in soil carbon (1.17 and 2.15% respectively). These figures are enormously important because they represent rare long-term records of changes in soil carbon after establishment of woodland within Europe (Smith, Powlson et al. 1997). As such they have been a crucial basis for calculating rates of soil carbon sequestration, and for validating modelling approaches to the effects of trees on soil organic matter content (Smith, Powlson et al. 1997), (Smith, Powlson et al. 2000), (Powlson, Smith et al. 1998). However, their accuracy and appropriateness for estimating the potential for soil carbon sequestration under bioenergy willow crops warrants careful consideration. First, simulations by the dynamic soil organic matter models RothC (Falloon, Smith et al. 1998) and CENTURY (Falloon, Smith et al. 1999) indicate that the Broadbalk figure for soil organic carbon is unrealistically high. As a consequence, the Smith *et al.* (2000b) study has utilised only the lower Geescroft Wilderness figure for calculations of potential rate of increase in soil carbon under tree bioenergy plantations. Note that it appears that this figure is slightly in error and should be 1.15% rather than 1.17%, since the time period over which the data was collected is reported as 102 rather than 100 years (Smith, Powlson et al. 1997). Second, the use of data on regeneration in natural woodland to directly estimate soil carbon sequestration under regularly coppiced and intensively managed plantations of monoculture sapling trees for biofuel production may result in errors. Effectively, the use of soil data from a natural regenerating woodland in this context assumes that regular harvesting of above-ground shoots has no impact on the sequestration of carbon below-ground. Clearly, experimental data on soil carbon under bioenergy coppiced tree plantations is urgently required, as well as alternative experimental approaches to estimating soil carbon changes.

##### 2.1.1.2 Total plantation area

The actual area planted with bioenergy trees is an important determinant of their overall contribution to mitigation of U.K. carbon emissions. The data calculations for potential carbon mitigation by bioenergy tree crops used in the U.K. study (Smith, Powlson et al. 2000) are based on the assumption that 10% of arable land (i.e. 658,900 ha) could be allocated to plantations. However, current U.K. goals are to achieve plantations covering a total of 125,000 ha by the target date of the initial Kyoto Protocol commitment period (2008) (DETR 2000). Thus, for this reason, the projections by Smith *et al.* (Smith, Powlson et al. 2000) probably overestimate the feasible carbon mitigation potential of bioenergy tree plantations for the U.K. within the first commitment period.

### 2.1.1.3 Depth of soil affected

In addition to total planted area, the depth of soil considered as appropriate in such calculations is critical. The Geescroft data relate to the uppermost 30 cm of soil and so this defines the depth limit of the above study (Smith, Powlson et al. 2000). However, short rotation coppice willow roots are known to extend to 50 cm by the first year (Welling 1996) and to greater than 1 m by the third year of plantation (Wyatt and Drake-Brockman 1995). Consequently, SRC willow plantations are likely to increase soil organic carbon contents down to a depth of at least 1 m in the soil profile. The effect of SRC willow plantations on soil organic carbon pools below 30 cm depth depends on the depth distribution of rates of soil carbon increase. However, if the rate of increase in soil organic carbon under SRC willow plantations is significant at depths greater than 30 cm, and the Geescroft Wilderness data are appropriate, then the calculations of Smith *et al.* (2000) underestimate the amount of carbon that would be sequestered on a per area basis in soils under bioenergy tree crops.

### 2.1.1.4 Initial soil carbon content

The first study of Smith *et al.* (2000b) assumes a decrease in soil organic carbon with depth in the soil profile down to 1 m, described by a negative exponential relationship indicating that only 13% of the total soil carbon is contained below 30 cm depth (Smith, Powlson et al. 2000). One consequence of assuming a steeply exponential decrease in soil carbon content with depth is that the carbon content of the surface soil can appear unusually high. For example, the arable land area in the U.K. is 6.589 M ha (Smith, Powlson et al. 2000). The total carbon content of that area is calculated on the basis of an exponentially decreasing relationship with depth to be 1140 Tg C to 30 cm depth (Smith, Powlson et al. 2000). If one assumes an average bulk density of 1.3 g cm<sup>-3</sup>, then this implies that the average carbon concentration in this upper soil horizon of arable land in the U.K. is 4.4%. Assuming similar bulk densities and soil depths, this concentration is substantially higher than the average soil carbon content for arable land across Europe (1.35%) (Smith, Powlson et al. 2000), or for the Geescroft Wilderness site prior to woodland regeneration (0.95%) (Smith, Powlson et al. 1997). The mean concentration of organic carbon in the arable soils of England and Wales was 3.3% (quarter percentiles 1.8% and 3.8%) in 1980, and 2.8% (quarter percentiles 1.7% and 3.1%) in 1995 (P. Loveland, *pers. comm.*). In a subsequent paper, (Smith, Milne et al. 2000) have revised their estimates of soil carbon sequestration using data indicating an average soil carbon concentration of 2.1% in the top 30 cm of arable soil in the U.K.

The extrapolations of Smith *et al.* (2000b) and (Smith, Milne et al. 2000) utilise the calculated *percentage rate increase* in soil organic carbon derived from the Geescroft Wilderness data (reported as a mean increase of 1.17% of initial soil carbon content per year) to determine soil organic sequestration rates under SRC willow plantations. This rate, when combined with the revised value for initial soil carbon concentration of potentially converted arable land cited above, is equivalent to an absolute rate of soil carbon sequestration of 0.98 Mg C ha<sup>-1</sup> y<sup>-1</sup>. This absolute rate of carbon sequestration to soil is higher than most observed rates for either SRC plantations (Table 1) or for natural woodland regeneration (Table 2). Extrapolations based on percentage (or relative) rate increases (rather than absolute rate increases) are extremely sensitive to the initial carbon content of the soil. Using this approach, an overestimate of the initial soil carbon concentration will result in an overestimation of the potential magnitude of carbon sequestered in soils under SRC. Furthermore, the use of relative rate increases to estimate potential soil carbon sequestration implicitly assumes that the higher the initial soil carbon concentration, the greater the potential sequestration. Several lines of evidence (Annex I: Sections 1.2.3.5 and 1.3) suggest that potential carbon sequestration may in fact be inversely related to initial soil carbon concentration.

### 2.1.1.5 Climatic and soil variability within the U.K.

The authors acknowledge that their estimates of the carbon mitigation potential for bioenergy tree coppice plantations do not take any account of the substantial variability in climatic and soil conditions within the U.K. (Smith, Powlson et al. 2000). Modelling approaches (Matthews 2000) and other European field studies (Makeschin 1999) suggest that local climatic and soil conditions are likely to be important determinants of SRC biomass yields. Without a detailed process-based model, it is not possible to accurately predict soil carbon sequestration underneath bioenergy crops in sites across the U.K.

### 2.1.1.6 Summary

In summary, the studies of Smith *et al.* (2000), (2000) on the potential for soil carbon sequestration under SRC willow are an extremely important contribution. They are based on the best available information, but the extrapolations presented have necessitated several major assumptions (see above). Clearly, appropriate, long-term, replicated SRC willow experimental data from a range of U.K. sites is required to evaluate the actual carbon mitigation potential of willow.

### 2.1.2 Experimental data from SRC trials in the U.K.

Although the carbon in harvested SRC willow crop material is removed from the site and eventually returned to the atmosphere, it is possible that non-harvested carbon may accumulate on plantation sites over relatively long time scales. Three such components may be identified: a) non-harvested above-ground biomass (coppice stumps); b) below-ground biomass (coarse and fine roots); and c) inputs to soil organic matter. The time-course of carbon accumulation in these components has been estimated on the basis of trial harvest data from 11 plots of willow and poplar that have been coppiced for varying time periods at three sites in the U.K. (Matthews, Robinson et al. 1994). Stump, coarse and fine biomass components all increased with time and were estimated to range from 3.6-6, 1.6-6.8 and 0.6-1.4 Mg C ha<sup>-1</sup> respectively after 25 years. Soil carbon

concentrations for the plots from one of the sites ranged from 2-4% C of total soil mass. Assuming that the soil was originally under arable farming practice and would have been expected to have a carbon concentration of between 1-3%, the authors concluded that there was little evidence of any change in soil carbon content as a result of SRC coppicing.

This study (Matthews, Robinson et al. 1994) is important because it reports data essential to evaluation of the soil carbon sequestration potential of SRC. Despite their obvious relevance to SRC carbon balance issues, it is very difficult to find such data in the SRC literature. However, the biomass accumulation time-course calculations are based on an extremely limited data set. In addition, the time span of the data is most likely too short to detect a significant change in soil carbon concentrations, given the resolution of most techniques to measure percentage organic carbon in the soil.

Finally, in evaluating the effect of SRC on soil carbon sequestration, it is critical to note that expressing the data as whole integers of the % carbon content of soil mass (e.g. Matthews, 1994) is unlikely to be useful. For example, an average increase of  $0.5 \text{ Mg C ha}^{-1} \text{ y}^{-1}$  in the total soil carbon concentration of a soil column of 0.5 m depth with a bulk density of  $1.3 \text{ g cm}^{-3}$  and an initial carbon content of 2.5% is equivalent to the storage of 0.625 Tg C over an area of 125,000 ha after 10 years. The equivalent change in soil carbon concentration over this period is from 2.5% to 2.58%. This example demonstrates that carbon concentrations and bulk densities will need to be determined to a high degree of resolution (at least two decimal places) in future research in order to adequately evaluate the potential for significant effects of SRC on soil carbon sequestration.

### 2.1.3 Current research in the U.K.

A network of 49 SRC willow and poplar plantation sites was established by the Forestry Commission between 1994/95 and 1995/96 (Armstrong 2000). A range of genotypes has been planted in each plantation. Sites were selected throughout the U.K. to cover the major soil types and a wide range of climatic conditions. At each site, soil physical and chemical properties (including particle size analysis, pH, cation exchange capacity, P, K,  $\text{NH}_4$  and  $\text{NO}_3$ , Mg, Ca and organic carbon) have been measured (Armstrong 2000). In an effort to emulate current SRC planting practice, sites were sprayed prior to ploughing with a contact herbicide, and planted with 25 cm unrooted cuttings. Residual herbicides were applied regularly to keep sites free of weeds. At the end of the first season, failures were replaced and all trees cut back to 10 cm high stumps to encourage multi-stemming. Harvesting is planned to follow a standard three year cutting cycle. Yield assessments are to be measured non-destructively each year (for modelling purposes). In addition, there will be regular monitoring of insect and disease impacts (Armstrong 2000). This network is likely to be a critically important resource in constructing yield models, determining best planting practices, identifying appropriate clone types, and in assessing effects of SRC tree plantations on carbon sequestration in U.K. soils. Data from the first rotation indicates that potential yields across all varieties for all sites varied by greater than a factor of almost 30 in years one and two, and by a factor of 15 in the year three (I. Tubby, Forestry Commission, *pers. comm.*). Recent data from the first harvest of the second rotation indicate that the range of yields across all varieties for all sites was also slightly greater than a factor of 30 (I. Tubby, Forestry Commission, *pers. comm.*). However, as yet there are no results available on the effects of these trial plantations on soil carbon content (I. Tubby, Forestry Commission, *pers. comm.*). Furthermore, plans for collecting data such as root biomass increments and changes to soil carbon in the next phase of the project (2002-2003) are restricted to a very limited number of sites, and are being carried out principally to inform yield models (S. Evans, Forestry Commission, *pers. comm.*). Nevertheless, the huge range of harvest yields across the sites suggests that carbon accumulation rates below-ground in the form of root biomass as well as sequestration rates in soil organic matter are likely to exhibit strong spatial heterogeneity.

### 2.1.4 U.S. research

Considerable research into the potential for SRC willow and poplar to mitigate  $\text{CO}_2$  emissions is ongoing in the U.S. Just as in the U.K., short rotation plantations have been established only since the 1980s – a short time period that restricts evaluation of the impacts of SRC on relatively slow processes such as soil carbon sequestration. Therefore, predictions have largely been developed based on principles of soil carbon balance, and data from analogous systems such as natural woodland succession on abandoned agricultural land (Grigal and Berguson 1998).

Nevertheless, two U.S. studies specifically address the issue of soil carbon sequestration under established SRC plantations. Soil carbon content was monitored in poplar plantations established on previously tilled agricultural prairie land, and compared to adjacent control grass and arable fields (Hansen 1993). Initial establishment and tending of the plantations resulted in a net loss of soil carbon over the first 6–12 years. This loss was largely from the surface 30 cm of soil, suggesting that it was due to enhanced decomposition. With time, soil organic carbon began to increase due to the input of leaf litter to the soil surface. In addition, canopy shading as well as the mulching effect of the litter build-up may have slowed soil decomposition by promoting cooler and wetter soil conditions. Over the full 18 years of the study, soil carbon to 1 m depth increased at an average rate of  $1.6 \text{ Mg ha}^{-1} \text{ y}^{-1}$  compared to control fields. This high rate of soil carbon sequestration may have been partly due to the inclusion of relatively deep soil horizons in the study. In particular, there was a substantial increase in the 30-50 cm soil layer that was attributed to tree root growth and associated carbon inputs to the soil.

A similar pattern of initial soil carbon losses was also suggested in a study of young poplar stands on grass and arable lands in Minnesota (Grigal and Berguson 1998). When the plantations were an average of 7-8 years old, there was no significant difference in soil carbon content between poplar and adjacent control fields. One of the factors that may contribute to early losses after plantation establishment is the initial carbon content of the soil. In the latter study, the previous agricultural practices of either row crops such as wheat with fertiliser or manure inputs, or pasture grass/hay resulted in soil carbon

contents averaging 3.75%. By contrast, intensively cultivated arable land tends to have lower soil carbon contents (see above) because persistent ploughing enhances soil carbon mineralisation (Paustian, Andren et al. 1997). Thus, relatively high organic matter inputs associated with specific agricultural practices prior to poplar plantation establishment may result in a greater proportion of labile soil carbon available for mineralisation during the bioenergy crop establishment phase.

Several other ongoing initiatives through the U.S. Department of Energy (see internet websites referenced at back) have investigated the potential for soil carbon sequestration with conversion of agricultural lands to biomass crops (hybrid poplar, cottonwood, sycamore, sweetgum, and switchgrass). In general, these studies have found that conversion from traditional tillage to biomass crops leads to increases in soil carbon storage (V. Tolbert, *pers. comm.*). Initial changes are in the surface soil layers with breakdown and incorporation of surface litter. As fine root turnover occurs with increasing age of the plantings (years 2+), soil carbon increases. The greatest changes occur on soils that have been in agriculture for longer periods of time or that have been subjected to substantial erosion and weathering resulting in relatively low soil organic matter contents (e.g., in the southeastern United States). In the upper mid-West (e.g., Minnesota, Wisconsin) where the organic matter content is high, temperatures are lower, erosion is low, and bulk densities are greater, soil carbon under bioenergy crops has increased, decreased, or stayed approximately the same over time depending upon the site characteristics.

Ongoing studies with willow systems for biomass production are focussed in the north-eastern U.S. (T. Volk, *pers. comm.*). To date, total soil carbon has not shown a consistent pattern of increase across the different willow sites examined. A recent detailed analysis of data from one site (a four year old willow plantation) indicated no change in soil carbon pools compared to adjacent field controls (Ulzen-Appiah, Briggs et al. 2000). The site was prepared for plantation without tillage (using chemical weed control), thereby possibly avoiding any initial flush of soil carbon mineralisation associated with soil turnover (Paul and Clark 1996). In addition, soil at the site had relatively high proportions of silt and clay particles. Fine silt (5-2  $\mu$ m) and coarse clay (2-0.2  $\mu$ m) fractions generally contain the highest concentrations of soil carbon because their high surface area enhances formation of organo-mineral complexes that protect soil carbon from microbial oxidation (Anderson and Paul 1984), (Post and Kwon 2000). The absence of initial tillage, and the high clay content at the site, may have inhibited the early net carbon release noted in other young bioenergy tree plantations (Hansen 1993), (Grigal and Berguson 1998). In considering heterogeneities within their data, Ulzen-Appiah *et al* (2000) estimated that 154 samples would be required to determine the whole soil carbon concentration to within 10% of the true mean in a 0.16 ha plot. A comparable study of switchgrass (*Panicum virgatum*) indicated that the smallest difference in soil carbon to 40 cm depth that could be significantly detected after five years was 1 Mg C ha<sup>-1</sup> (Garten and Wulschleger 1999). Initial soil carbon concentrations at the sites in this study generally ranged from 0.5-1.0%. Thus, the change is equivalent to an *increase* in the initial soil carbon concentration of 2-3% (or an absolute change in soil carbon concentration from 0.5-1% initially to 0.52-1.02% at the end). Analyses of the variability in the site data indicate that statistically significant detection of this change would require a sample size in excess of 100 (Garten and Wulschleger 1999). In conclusion, the current U.S. experience is that there is substantial variability in the potential for soil carbon sequestration under a wide range of bioenergy crops because of heterogeneities within sites as well as larger scale variations in soil type, land-use history, climate, and crop productivity (V. Tolbert, T. Volk, *pers. comm.*).

### 2.1.5 Other European research

Across northern Europe, there are several countries actively pursuing research into SRC willow and poplar as a source of renewable bioenergy. For example, Sweden has investigated many aspects of the establishment and utilisation of bioenergy short rotation forest trees (Verwijst 2000). However, studies investigating potential impacts of bioenergy trees on soil carbon content have been initiated only in recent years (Börje Alriksson, *pers. comm.*). In a study of a six year old mixed stand of willow, poplar and alder, decomposition of soil organic matter exceeded carbon inputs from the trees resulting in a net carbon loss from the system (Börje Alriksson, *pers. comm.*). These data support the conclusions drawn from studies on young poplar plantations in the U.S. (Hansen 1993), (Grigal and Berguson 1998) (see above). In a study of three mixed tree coppice plantation sites across Germany, substantial increases in soil carbon in the upper surface soil were observed after seven to nine years ((Jug, Makeschin et al. 1999), Table 1). This increase was attributed to leaf and root litter inputs from tree and ground vegetation to the surface soil horizon (Jug, Makeschin et al. 1999). However, the soil organic carbon content of deeper soil horizons (10-30 cm) at one of the sites was reduced by a similar rate resulting in no net change in total soil carbon to 30 cm depth.

In summary, the limited data available so far indicate that SRC bioenergy tree plantations can have a wide range of effects on soil carbon contents. The state of our current experimental knowledge on the impact of bioenergy tree crops on soil carbon sequestration is summarised in Table 1. According to the studies available, soil carbon under bioenergy crops can be increased, unchanged, or reduced. Substantial further research is required, as well as longer elapsed time since plantation establishment, in order to fully evaluate the spatial and temporal patterns of effects of bioenergy trees on soil carbon sequestration.

**Table 1. Rates of soil carbon sequestration since establishment of short rotation coppice plantations.**

Site history	Years since plantation establishment	Soil sample depth (cm)	Average rate of soil organic carbon increase (kg C ha <sup>-1</sup> y <sup>-1</sup> )	Reference
Agriculture to short	12-18	100	1630	(Hansen 1993)

rotation coppice poplar (U.S.A.)				
Agriculture to short rotation coppice poplar (U.S.A.)	6-15	0-25 0-100	<i>No change</i> <i>No change</i>	(Grigal and Berguson 1998)
Grass/shrub land to short rotation coppice willow (U.S.A.)	4	0-60	<i>No change</i>	(Ulzen-Appiah, Briggs et al. 2000)
Agriculture to short rotation coppice mixed poplar, aspen and willow (Germany)	7-9	0-10 10-30	100-555 <i>gain</i> 0-555 <i>loss</i>	(Jug, Makeschin et al. 1999)

### 2.1.6 Indications from analogous natural woodland regeneration sites

In addition to experimental trials and modelling exercises, research into the potential for soil carbon sequestration has utilised documented cases of natural woodland regeneration on previously agricultural land to provide an indication of the likely boundaries on sequestration rates. Studies of secondary succession in temperate latitudes indicate a wide range of rates of soil organic carbon increase (Table 2). Poulton's data (1996) that were used by Smith *et al.*, (2000) as a basis to derive potential soil carbon sequestration rates under SRC willow in the U.K. have been included at the bottom of the table for comparison. Clearly, the basic rate of increase of soil carbon under natural woodland regeneration used by Smith *et al* is well within range of other studies.

**Table 2. Rates of soil carbon sequestration during re-establishment of deciduous forest after agricultural use in cool, temperate moist latitudes (adapted from (Post and Kwon 2000)).**

Site history	Years since agriculture	Soil sample depth (cm)	Average rate of soil organic carbon increase (kg C ha <sup>-1</sup> y <sup>-1</sup> )	Reference
Old field succession to hardwoods (U.S.)	1-60	10	151	(Zak, Grugal et al. 1990)
Old field succession to mixed oak (U.S.)	>250	15	94	(Robertson and Vitousek 1981)
Old field succession to hardwoods (U.S.)	>100	10	116	(Robertson and Tiedje 1984)
Old field to managed pine plantation (U.S.)	10-50	15.4	657	(Wilde 1964)
Abandoned field to mixed forest (U.S.)	2-66	43	22	(Hamburg 1984)
Constructed dike to forest (Canada)	100	12	263	(Beke 1990)
Hardwood plantation (U.K.)	20-21	60-70 (mineral soil)	537	(Ovington 1956)
Agriculture to oak forest (Broadbalk U.K.)	100	30	561	(Jenkinson 1990)
Agriculture to oak forest (Geescroft, U.K.)	102	30	426	(Poulton 1996)

### 2.1.7 Summary

Thus, the data from tree bioenergy plantations and natural woodland regeneration sites so far indicate that:

- there may be an initial net loss of soil carbon during the plantation establishment phase. The extent of this loss will be dependent on previous land use history and its effect on the extent of labile (easily mineralisable) soil carbon pools.
- trial plantations have not been established for a sufficient time to determine with confidence the occurrence of significant changes in soil carbon. However, early indications are that where soil carbon has increased in short rotation coppice plantations (Table 1), the rate of increase exceeds the range of the soil carbon increases observed with natural woodland regeneration on previously agricultural land (Table 2).
- there is likely to be substantial variation in soil carbon sequestration rates between sites due to variations in soil type, land-use history, climate, topography/drainage characteristics and crop productivity.

## 2.2 Miscanthus

### 2.2.1 Published research from the U.K.

Most research so far on miscanthus in the U.K. has centred on quantifying its yield potential, medium/long-term viability as a suitable energy crop, and its physiological response to U.K. temperatures (Bullard 1999). To date, there is little information on effects of miscanthus plantations on soil carbon status in the U.K. (M. Bullard, J. King, *pers. comms.*).

### 2.2.2 U.K. current research

Plantations of *Miscanthus sacchiflorus* were established by ADAS (Agricultural Development and Advisory Service) across seven sites in the U.K. from 1992-1994 (Bullard 1999). Sites were selected across a strong precipitation gradient in order to test the effect of soil moisture regime on yield. In subsequent harvests, two components of growth of miscanthus plantations were identified: a yield building phase (2-5 y); and a subsequent equilibrium phase during which yields of stem material are maximised and ranged from 5.2-7.2 Mg C ha<sup>-1</sup> y<sup>-1</sup> (Bullard 1999). Yield variation between sites was principally due to growth limitation by summer drought, and low temperatures that constrain the length of the growing season. Although no specific data on changes in soil carbon content were reported, the high leaf litter production (1.2-2.8 Mg C ha<sup>-1</sup> y<sup>-1</sup>) indicates a substantial carbon input onto the soil surface, as well as a build-up of mulch that may inhibit decomposition. Together, these observations suggest that there may be considerable potential for enhanced formation of soil organic matter under miscanthus plantations. Indeed, a very recent report indicates rates of soil organic carbon sequestration under miscanthus of c. 500 kg C ha<sup>-1</sup> y<sup>-1</sup> (Bullard and Metcalfe 2000). Miscanthus trial plantations have also been established at IACR Rothamsted.

### 2.2.3 Other European research

Miscanthus field trials have been established in many other European countries since 1983 in order to support research on propagation and establishment, management practices, and crop harvesting and handling (Lewandowski, Clifton-Brown et al. 2000). However, as yet there have been very few investigations of the effects of these plantations on soil carbon (E. Hansen, J. Clifton-Brown, *pers. comms.*). In Germany, after 6-8 years of *Miscanthus X giganteus* cropping, soil organic matter to 40 cm depth was reported to have increased by an additional 0.2% on silty soils, and 0.5% on sandy soils (Beuch, Boelcke et al. 2000). Assuming a bulk density of 1.3 g cm<sup>-3</sup>, an initial soil carbon concentration of 2%, these increases amount to additions of 208 and 520 kg C ha<sup>-1</sup> y<sup>-1</sup> respectively. Several other European researchers either have plans to measure, or are in the process of analysing, changes in soil carbon under miscanthus, but as yet the data are not available (E. Hansen, J. Clifton-Brown, *pers. comms.*). A recent study of miscanthus root distribution recorded that half of the total root biomass was below 90 cm, and that some roots were present down to the maximum depth measured of 250 cm (Neukirchen, Himken et al. 1999). These results suggest that miscanthus could have stronger effects than short rotation coppice bioenergy trees on soil properties in relatively deep soils horizons.

### 2.2.4 Indications from natural grassland regeneration sites

Several studies indicate relatively high soil carbon sequestration rates when previously cultivated arable land is converted to permanent pasture (Table 3). However, there is considerable controversy over the potential of perennial grassland to sequester soil carbon. A recent study suggests that soil carbon in land under permanent pasture in the U.K. has decreased over the past 15 years (Loveland, Webb et al. 2000). The rates of soil carbon sequestration in Table 3 below are intended as indicators of the highest observed soil carbon sequestration rates on previously cultivated arable land that has been converted to permanent pasture. These data are indicative of potential soil carbon sequestration rates when tillage is ceased, and the vegetation is subjected to regular grazing and/or mowing each year. If such conversions to permanent natural grassland are analogous to establishment and harvesting practices in miscanthus plantations, then the rates in Table 3 may be indicative of the range of soil carbon sequestration rates under miscanthus in the U.K.

**Table 3. Rates of soil carbon sequestration during re-establishment of perennial grasslands after agricultural cultivation in cool, temperate moist latitudes (adapted from Paustian *et al.*, 1997)**

Site history	Years since agriculture	Soil sample depth (cm)	Average rate of soil organic carbon increase (kg C ha <sup>-1</sup> y <sup>-1</sup> )	Reference
Planted grassland (New Zealand)	18	20	1000 (assuming a bulk density of 1.3 g cm <sup>-3</sup> )	(Haynes, Swift et al. 1991)
Abandoned field to grassland (U.K.)	83	23	550	(Jenkinson 1971)
Planted pasture grassland (U.K.)	30	15	600	(Tyson, Roberts et al. 1990)

### 3 Review of existing SOM/carbon models

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Several models of soil organic matter were evaluated against long-term datasets at a NATO Advanced Research Workshop held at Rothamsted in May 1995 as part of the Global Change and Terrestrial Ecosystems (GCTE) Soil Organic Matter Network (SOMNET). The models were all process-oriented multi-compartment models, although decomposition processes were generally described empirically. Most of the models use similar compartmentalisation of soil organic matter into functional pools which decay according to first-order kinetics with differing rate constants, and often include an inert organic matter compartment.

We have summarised below the descriptions and performance of these models for this report.

#### 3.1 ITE (Edinburgh) Forest and Hurley Pasture ecosystem models

The Edinburgh Forest (EF) model and the Hurley Pasture (HP) model share many of the same sub-models, and are therefore discussed together. The EF model is described in detail by Thornley & Cannell (1992), and the HP model by Thornley & Verberne (1989). In the EF model, trees are divided into foliage, branch, stem, coarse root and fine root compartments, while in the HP model, grass is divided into four different age categories of root and leaf. In both models, flows of carbon, nitrogen and water between adjacent compartments occur in response to concentration or water potential gradients across resistances determined by compartment size. Structural growth of the various compartments is governed by local substrate concentrations and temperature. The models operate on a 15 min time-step to take into account rapid changes in photosynthesis. In the original versions of the models, the soil is treated as a single homogenous layer, with pools of SOM carbon and nitrogen, and pools of  $\text{NH}_4^+$  and  $\text{NO}_3^-$ . These last two pools are influenced by atmospheric deposition, N-fixation, fertiliser application, plant uptake, volatilisation, nitrification, denitrification, and leaching. Carbon is cycled between living biomass (trees or grass) with a respiration loss on each transformation. Nitrogen cycling is similar, but additionally a 'local' N pool is defined which is in equilibrium with the  $\text{NH}_4^+$  and  $\text{NO}_3^-$  pools, and to which the biomass has 'privileged access'. The rates of all microbial transformations depend on the size of the biomass and substrate pools, the temperature and moisture potential. Plant N uptake is determined by the size of the fine root compartment, plant nutrient status, the soil mineral N level, and the temperature and water potential of the roots. Soil moisture potential is determined from the soil water content using a simple power-law approximation to the moisture release curve.

During an evaluation of these two models (Arah, Thornley et al. 1997), it was found necessary to revise the SOM sub-model described above. In the new version, SOM exists in three discrete fractions, 'unprotected', 'protected', and 'stabilised'. These pools respectively have C/N ratios of 15, 10, and a soil texture-dependent value between 17 and 20. Movements of C from the unprotected and protected pools to the stabilised pool follow first-order kinetics relations with standard rate constants of  $2 \times 10^{-4} \text{ d}^{-1}$  for the former, and  $2 \times 10^{-5} \text{ d}^{-1}$  for the latter. These rates can be modified by the same temperature and water status multipliers used in the first version of the model. Each pool is also subject to mineralisation with different rate constants, producing soluble C and  $\text{NH}_4^+$ . This  $\text{NH}_4^+$  can be nitrified to  $\text{NO}_3^-$  which is subject to denitrification, leaching and plant uptake. Soluble C is also produced by root exudation, and can be lost by leaching. Biomass has a fixed C/N ratio of 8, growth depends on the concentration of soluble C,  $\text{NH}_4^+$  and  $\text{NO}_3^-$ , temperature and soil water status. Fresh organic matter pools (from leaf and root senescence, and from biomass death) are divided into metabolic, cellulose and lignified fractions, the latter two with fixed C/N ratios of 150 and 100, respectively. Decomposition of the metabolic and cellulose fractions produces soluble C and  $\text{NH}_4^+$ , while that of the lignified fraction produces unprotected and protected SOM in a ratio determined by the clay content of the soil.

In the evaluation of the two models against long-term soil carbon data (Arah, Thornley et al. 1997), it was found that in the EF model the trees died after about 10 years because of a low plant water status. Inactivation of the water balance sub-model allowed soil C dynamics under forest to be predicted with reasonable accuracy, but N dynamics were generally overestimated due to leaching not being taken into account. Similar problems with the soil water balance sub-model were experienced in the HP model, but even with inactivation of this, it performed poorly in predicting SOM changes under grassland.

#### 3.2 RothC-26.3

RothC-26.3 is the current Rothamsted model for the turn-over of organic carbon in soil, and has been described by Coleman & Jenkinson (1996). It is solely concerned with soil processes, and does not contain a sub-model for plant production, and does not attempt to calculate annual returns of plant C to the soil from above-ground yields. Turnover of C is sensitive to soil type, temperature, moisture, and plant cover. Nitrogen and C dynamics are not interconnected, the inert organic matter compartment is initialised using carbon-dating, and starting values are obtained by running the model to steady state. It uses primarily monthly data, and shares several basic concepts with CENTURY. RothC is applicable to freely-drained mineral soils, and not to permanently waterlogged or organic soils.

RothC-26.3 operates in two modes. If the organic carbon inputs are known, it can be run in 'forward mode' to calculate how this input will decay in a particular soil under a particular climate, and how this influences soil organic carbon. However, generally, these organic carbon inputs (i.e. from litter decay, root death, root exudation, turnover of microbial biomass, etc.) are not accurately known, and the model can, therefore, be run in 'reverse' mode to calculate what these inputs must be in order to match an observed change in soil organic carbon levels.

Coleman *et al.* (1997) describe the testing of RothC-26.3 against 18 long-term soil organic carbon data-sets. There was acceptable agreement between observed and predicted data for 14 of the data-sets.

### 3.3 DAISY

The DAISY model is a process model simulating crop production, soil water and N dynamics in agroecosystems, and was not designed specifically for C dynamics (Hansen, Jensen *et al.* 1991). Nevertheless, soil C pools and turnover rates are necessarily taken into account due to the close association between soil C and N dynamics. Parameterisation of the SOM sub-model has been from both long-term experimental data and short-term laboratory incubation data. The model consists of sub-models for soil water dynamics, soil temperature, SOM turnover, soil mineral N transformations, crop growth and N uptake, and system management (tillage, harvest, etc.). It was designed primarily for use at the plot or field level, but has also been used at catchment (Styczen and Storm 1993) and regional (Jensen, Stougaard *et al.* 1994) scales. The SOM module consists of three main pools – fresh organic matter, soil microbial biomass, and native soil organic matter, each of which have two sub-pools representing slower and faster rates of turnover. Decomposition of each sub-pool is described by first-order kinetic equations, the rate coefficients of which are derived from laboratory incubations, and are influenced by soil water content, soil temperature, and for some of the pools, soil clay content. The effect of low pH on these decomposition rates is not simulated. A semi-cohort accounting system is used for litter decay. Production of CO<sub>2</sub> results from all C fluxes into the soil microbial biomass pools (i.e. substrate utilisation efficiencies are less than unity) and from microbial respiration. As the flows of material in the SOM model are based on C, the corresponding N pools are calculated from the amount of C in each pool using fixed C/N ratios for each pool. Net N-mineralisation or N-immobilisation is derived from the N balance. The model includes NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup> pools, and simulates the processes of nitrification, denitrification, and leaching losses. Input data requirements include daily values of solar radiation, air temperature and precipitation, as well as soil water characteristics such as the water release parameters and hydraulic conductivity. System management information, such as the amount and timing of organic additions, must also be specified.

DAISY is not really applicable to forest systems as it is currently not able to simulate tree growth and the associated water use and microclimate (i.e. canopy rainfall interception, shading, soil temperature). It also does not include litter-fall from the canopy, organic matter only being added to the soil through crop root decomposition at the end of the season or application of crop residues or manures. Dead root turnover or rhizodeposition during the season (which may be around 1 t C ha<sup>-1</sup> y<sup>-1</sup>, Jensen, Mueller *et al.* 1997) are also not simulated, neither is leaching of dissolved organic C and incorporation of plant litter by micro-organisms accounted for. In pastures, the N returned to the soil in the excreta and urine of any grazing animals is assumed to be 40% of pasture yields. Ploughing of pasture results in the incorporation of both roots and above-ground biomass.

The model has been validated on its ability to simulate short-term nitrate leaching, crop production and crop N uptake, i.e. within a single growing season or crop rotation {e.g. \Hansen, 1991 #495}, and also against long-term soil organic matter data-sets (Jensen, Mueller *et al.* 1997). In the latter, it performed reasonably well in simulating trends in total soil C levels at most of the arable sites, but could only simulate changes in a grassland site by making assumptions about the plant-derived C inputs to the soil. Nevertheless, out of the nine models tested in this comparison, DAISY performed the best in terms of agreement using the RMSE (root mean square error) statistic across all the observed datasets (Smith, Smith *et al.* 1997).

### 3.4 CENTURY

The CENTURY model simulates the dynamics of C, N, P and S in terrestrial ecosystems to a soil depth of 20 cm (Parton, Scurlock *et al.* 1993). Soil organic matter is subdivided into three pools (active, slow and passive) based on their decomposition rates, and litter is subdivided into two pools (metabolic and structural). Carbon leaving the active SOM pool enters the slow SOM pool, with a proportion being lost as CO<sub>2</sub>, the magnitude of which is determined by soil texture. Soil texture also influences the rate of transfer between the slow and passive pools. Plant growth sub-models include grass/crop, forest, and savannah. The model runs on a monthly time-step, and requires monthly average minimum and maximum air temperatures and rainfall as input. The model is only sensitive to soil pH when phosphorus is being considered, and does not take into account the decrease in decomposition rates at low pH values, such as can occur with the development of a litter layer. Vegetation in the CENTURY model is a function of the potential primary production, a temperature curve, and the C/N ratios and lignin contents of biomass pools. Default crop parameterisation is provided with the model. CENTURY has been used to simulate C sequestration during soil formation (Parton, Schimel *et al.* 1987), and changes in soil C storage under different climate change scenarios (Schimel, Braswell *et al.* 1994).

CENTURY was evaluated against seven long-term data-sets from different climates and crop management scenarios (Kelly, Parton *et al.* 1997), and was able to simulate changes in soil C and N reasonably accurately, but was more successful at simulating SOM in grass and crop systems than forest systems. This is perhaps not surprising, as the model was developed for, and most widely parameterised and tested for, temperate grasslands and agroecosystems, with the forest sub-model being less well tested than the grass and crop sub-models. Most of the differences between different scenarios were due to differences in the slow C pool. CENTURY was one of three models out of the nine evaluated that did not show a significant bias over all simulations (Smith, Smith *et al.* 1997). In particular, the explicit linkage between C and N dynamics in the model allowed it to capture differences between low- and high-N datasets.

The structure of CENTURY makes it impossible to accurately simulate the sequestration of SOM in a forest with a developed litter layer, as in reality, this highly decomposed layer remains on the surface of the soil, but in the model is transferred to a pool of slow SOM, giving unrealistically high values of SOM. This limits its usefulness to simulate SOM changes in forested systems, although it was successful in modelling tree biomass pools. Plans are underway to improve the model structure to incorporate this litter layer to make it more applicable to forest systems (Kelly, Parton et al. 1997). Similarly, it had limited success in predicting inter-annual variability in yield and N offtake values, which may have been due to inaccuracies in the estimated minimum and maximum temperature data used due to lack of observed data. Prediction of annual N offtake was more accurate than annual yield.

### 3.5 DNDC

The DNDC (DeNitrification and DeComposition) model simulates carbon and nitrogen cycling in agroecosystems at a daily or sub-daily time-step (Li, Frohking et al. 1994). There are four main sub-models – soil physical environment, plant growth, organic matter decomposition, and denitrification. The soil physical environment module uses soil physical parameters, daily weather, and plant water use to determine soil temperature and water profiles. The plant growth module simulates daily water and nitrogen uptake by vegetation, root respiration, plant growth and partitioning of biomass, and requires potential crop yield as an input. The decomposition module calculates organic matter decomposition, nitrification, volatilisation, and CO<sub>2</sub> production on a daily time step. The denitrification module operates at an hourly time-step to simulate the production of nitric oxide, nitrous oxide, and dinitrogen gas.

Thus, DNDC requires as input: (1) daily air temperature and rainfall; (2) soil bulk density, texture or clay content, pH, and initial C content of the surface soil; and (3) information on crop management operations such as crop rotation, tillage, fertilisation, manure application, and irrigation.

The initial soil C content is partitioned into the various pools in the model according to default coefficients: litter (15%), microbial biomass+active humus (15%) and passive humus (70%), although these coefficients can be changed if necessary. Clay adsorption of humus allows some site-specificity. When the crop is harvested, the model assumes that all of the root biomass and a specified fraction of the stem biomass is added to the litter pool.

DNDC was evaluated against eleven long-term data-sets (Li, Frohking et al. 1997), and was able to capture the overall behaviour of the soil organic carbon % in all but one of the data-sets. In particular, it successfully simulated the decline in SOC% from around 2.7% to about 1% after conversion of grassland to cultivated cropping.

### 3.6 NCSOIL and NCSWAP

The NCSOIL model (Molina 1996) simulates C and N transformations in soils, and is built around the concept of potentially mineralisable nitrogen. The soil organic matter is divided into four pools which decompose according to first-order kinetics with differing rate constants. Two of these pools represent microbial biomass, a third pool the remaining mineralisable N, and the fourth pool the slowly decomposing organic matter. Residues are described by a further two pools. Decay rates may also be influenced by water, temperature, clay content and N status of the soil. Flows of N and C are interconnected, and increasing stability of organic matter results from metabolism and not from sorption mechanisms that would be sensitive to clay content. In the original version, microbial succession is simulated on residues, and although decomposition rate is independent of microbial biomass, microbial succession leads to more stable materials. Since then, however, the model has been simplified to avoid having to determine too many initial variables and parameters, by collapsing microbial succession into one microbial compartment whose dynamics are determined by the rate of flow of C through populations that consume microbes (Molina 1996).

Although NCSOIL is a stand-alone model, it also is part of the larger NCSWAP model (Clay, Clapp et al. 1985) which calculates C and N transformations in the soil-water-air-plant system. NCSWAP requires as inputs the initial soil conditions, details of crops grown, temperature, rainfall, and crop management information. The model operates on a daily time-step, but C and N transformations are calculated five times daily if there is water infiltration.

In the evaluation against long-term datasets (Molina, Crocker et al. 1997), NCSWAP generally simulated above-ground production well, and gave close agreement with soil organic carbon levels under grassland, arable cropping, and woodland. It also was able to simulate well the decline in soil organic matter after the conversion of virgin grassland to arable cropping.

### 3.7 CANDY

CANDY (CARbon and Nitrogen DYNAMICS) is a modular system containing different sub-models and a database system for model parameters, measurement values, initial values, weather data, and soil management data. The model is based on long-term experiments at Bad Lauchstädt in Germany, and is used for simulating the short-term dynamics of nitrogen transformation and the long-term dynamics of carbon sequestration in arable soils. A detailed description and initial evaluation of the model is given in Franko *et al.* (1995), and further validation on a number of long-term datasets is given by Franko *et al.* (1997). Use of the model for long-term simulations is in two parts. The basic model operates on a daily time step and calculates a measure of the turnover activity, or 'biological active time' (BAT), and the flux of carbon from fresh OM to SOM, termed 'reproducing carbon' (Franko, Komarov et al. 1997). Organic matter inputs to the soil are provided as inputs to the model, with different types (e.g. straw, slurry, manure, compost) being accommodated. Inputs from crop residues are

calculated from crop yields by linear regressions. The daily model requires a detailed soil description, daily weather data (solar radiation, air temperature and rainfall), crop yields, and an initial value for the decomposable carbon pool.

A second simpler model, operating on an annual time step, and implemented in an Excel spreadsheet, uses the BAT and the annual flux of reproducing carbon calculated with the daily model. Nitrogen fluxes are not simulated. This model also requires an initial value for the decomposable carbon pool. The goal-seeking facility of the spreadsheet can be used for parameter estimation.

Both models require an estimate of inert carbon, which can be either estimated by optimisation or obtained from experimental measurements.

### 3.8 SOMM

The SOMM model (Chertov and Komarov 1997), developed in Russia, is designed to simulate soil organic matter dynamics under forested systems, taking into account the three types of humus, mor, moder and mull. Three compartments are defined - (a) undecomposed litter, (b) partially decomposed litter, and (c) humus in the mineral topsoil. Six processes of mineralisation and humification by three groups of micro-organisms are considered, which can be influenced by C/N ratio, temperature and water status of the soil. Different types of humus (mor, moder and mull) reflect proportions of earthworms in contrast to micro-arthropods, etc. The structure and calibration of SOMM is from laboratory data only. Input data required include total litter mass (crop and root residues), nitrogen and ash contents of the litter, information on soil water status (daily or monthly), soil temperature, and the initial SOM and N contents in each of the three compartments.

In the comparison against long-term soil organic matter datasets (Chertov, Komarov et al. 1997), SOMM performance was generally poor, with a tendency to underestimate soil organic matter levels for a number of the datasets used, and to overestimate SOM for others. However, this may have been because several of the datasets were for grassland and arable cropping systems, whereas SOMM was developed for forest systems. Other weaknesses identified for the model included the fact that (1) SOM transformation rates are not dependent on soil properties, but only the N and ash content of the fresh litter and the C/N ratio of the stable humus; (2) the soil is treated as a single layer; (3) the partially humified litter pool does not distinguish between humified forest floor material and 'labile humus' in the mineral top soil; (4) calculations are on SOM rather than carbon; and (5) unrealistic values for some of the rate variables.

### 3.9 Verberne/MOTOR

The Verberne model in its original form was designed to simulate N and water balance in a grassland-soil-plant system in order to predict yield, N-uptake, N leaching, N mineralisation and accumulation of soil organic N, and is described in detail by Verberne *et al.* (1990). The grass growth sub-model is based on the ITE model described above (Thornley and Verberne 1989). The soil is simulated using three sub-models: (1) a soil water sub-model, (2) a soil organic matter sub-model, and (3) a soil N sub-model. The soil water sub-model used was the SAWAH water balance model (ten Berge, Jansen et al. 1992), but this requires a large amount of data, making it difficult to use with many datasets. For this reason, a simpler version of the model was developed by programming the Verberne model in the MOTOR (MODular description of the TurnOver of organic matter) system (Whitmore, Klein-Gunnewiek et al. 1997).

The structure of the Verberne/MOTOR model is similar in many respects to other models. Fresh organic matter from plant material is subdivided into three pools: (a) decomposable plant material, (b) structural plant material, and (c) resistant plant material, with the amounts decomposing from each pool according to first-order kinetics with an appropriate rate term. Distribution of fresh organic matter between these three pools is made on the basis of its C/N ratio. The decomposed carbon from the first two of these pools can be taken up by either unprotected or protected microbial biomass pools. These, in turn, contribute correspondingly to unprotected or protected organic matter pools, and from there, to the stabilised soil organic matter pool. The resistant plant material pool contributes directly to the protected organic matter pool without passing through microbial biomass first. There are, therefore, three SOM pools. In all cases, flux rates are assumed to be functions of the size of the associated source pools, modified by soil moisture and temperature. Protected organic matter refers to physical protection caused by soil clay, and decays at a rate 100 times slower than unprotected organic matter.

Unlike most of the other models, Verberne/MOTOR assumes a linear relationship for the temperature multiplier in the range 0-25°C, based on data given by van der Linden *et al.* (1987). Similarly, the multiplier for soil water status is assumed linear between 85% and 10% of available water (van Keulen and Seligman 1987), and declines only slowly as the soil dries beyond 10%.

In the evaluation against long-term datasets (Whitmore, Klein-Gunnewiek et al. 1997), the Verberne/MOTOR model performed well, with 48% of the simulations within  $\pm 5\%$  of the corresponding measurement, and 86% within  $\pm 10\%$ .

### 3.10 Hybrid

Hybrid v3.0 (Friend, Stevens et al. 1997) is another model developed at ITE Edinburgh, but is based on quite different concepts to the Edinburgh Forest Model discussed above. Hybrid describes the cycling of carbon, nitrogen and water within a specified terrestrial ecosystem consisting of trees of varying heights and characteristics, and grass. Carbon assimilation, and water and nitrogen uptake in the trees are simulated on a daily time-step, with the assimilated carbon being stored in a pool

which is distributed to the tree components at the end of each year. The growth of grass is followed on a daily time-step. Individual trees and grass compete with each other for light, water and nitrogen within a 'plot'. Competition for light results in shading of smaller plants with larger ones. Larger plants also take up a greater proportion of available water and nitrogen. The initial size of each tree seedling may be determined stochastically; mean behaviour is predicted by simulating a number of plots and calculating their averages. Early versions of the model were one-dimensional - i.e. the combined canopy was divided into a number of horizontal layers for calculation of light interception, and the soil was represented as a single layer. Subsequent versions, however, have disaggregated the canopy to handle individual trees of varying sizes, and routines describing the dynamics of soil carbon and other nutrients have been taken directly from the CENTURY model with minor modifications for use in ecosystem carbon balance studies.

Hybrid was not one of the models that was evaluated with the long-term soil carbon datasets, so it is not possible to say for certain how well it performs over these periods. However, since the soil carbon routines are almost identical to those of CENTURY, it is likely that it performs in a similar way. Any discrepancies are likely to be due to errors in the estimation of C inputs from the vegetation into the system.

### 3.11 ICBM

ICBM, the Introductory Carbon Balance Model, was developed in Sweden describing carbon dynamics under arable crops (Andren and Katterer 1997). It consists of two compartments representing young and old carbon, and four fluxes (governed by four rate-determining parameters), and one parameter,  $r_e$ , combining most external factors affecting C mineralisation (temperature, precipitation, drainage, etc.). Parameters for litter input, rate of humification, and various external influences are also incorporated. Because the model is very simple, the differential equations could be solved analytically, and the values of the parameters can be estimated using nonlinear regression programmes. Sensitivity showed that initial soil C mass, current C inputs, and soil temperature and moisture level, were the most important factors determining whether C stocks declined or increased. The model was calibrated using data from a 35-year experiment with arable crops on a clay soil in central Sweden, and was able to simulate the changes in the two soil carbon pools in response to changed inputs, climate, initial pools, litter quality, etc., over 30 years. ICBM was subsequently used to analyse topsoil carbon data collected from a number of long-term field trials in Sweden and other Northern European countries (Katterer and Andren 1999).

As ICBM was not one of the models that was evaluated with the long-term soil carbon datasets, it is not possible to say how well it performs over these periods.

### 3.12 General comments

The comparison of nine soil organic models against long-term datasets described by Smith, Smith *et al.* (1997) showed that the models fell into two groups in terms of overall performance. The group containing SOMM, ITE and Verberne performed significantly poorer than the group containing RothC, CANDY, DNDC, CENTURY, DAISY and NCSOIL. Some of this difference could be ascribed to the level of site-specific calibration used – some models took advantage of this, while others did not. The other main factor was that all of the models in the poorer performing group were less developed for land-use systems other than that for which they were developed in the first place. ITE and SOMM, for example, are forestry/grassland models and attempted to simulate arable crops assuming they were grass. These two models did perform better on forestry datasets, but interestingly, no better than other more generally applicable models.

Characteristics of the nine models evaluated by Smith, Smith *et al.* (1997) and the other two models reviewed are summarised in Table 4. DAISY performed best across all the datasets, but is designed specifically for arable crop ecosystems. RothC also performed well in terms of simulating soil carbon processes, although a limitation of this model is that it does not simulate C inputs into the system. Specific routines describing stand growth would need to be incorporated in order for it to be useful in the short-rotation coppice context. In terms of general applicability and to forest systems in particular, the CENTURY model (or at least its routines) would appear to be the most suitable model to use as a basis to evaluate carbon sequestration by short-rotation coppice systems. The Hybrid model may also be suitable for use in this context, but does not appear to have been so widely validated.

The U.S. Department of Agriculture have recently announced that they are developing a model termed 'CQESTER' to predict changes in soil carbon based on site-specific farming management practices (see final website in Annex II). The model has been designed to estimate the short-term soil carbon gain or loss each year that may arise from switching between various tillage practices. As such, this model may have considerable potential for adaptation to predicting the effects of bio-energy crops on soil carbon sequestration.

**Table 4: Summary of characteristics of various SOM models and their performance in relation to predicting long-term changes in soil C pools**

Model name	Country of origin	Land-use systems designed for	Time-step	Input data	Comments	Performance
CANDY	Germany	Arable	Daily	Soil description, daily solar radiation, air temperature and rainfall, crop yields, initial values for decomposable C and inert C pools.		✓✓✓
CENTURY	USA	Grasslands, arable, forest, savannah	Monthly	Maximum and minimum temperatures, rainfall.	Includes 2 litter pools, 3 SOM pools	✓✓✓
DAISY	Denmark	Arable		Daily solar radiation, air temperature and precipitation. Soil water release parameters and hydraulic conductivity. System management information (amount and timing of organic additions).	Designed primarily for N, but includes C. Six OM pools. Used at field, catchment, and regional levels.	✓✓✓✓
DNDC	USA	Arable	Daily	Daily air temperature and rainfall; soil bulk density, texture or clay content, pH, and initial C content of the surface soil; information on crop management (i.e. crop rotation, tillage, fertilisation, manure application, and irrigation).	3 OM pools: litter, microbial biomass+active humus, passive humus. Used to predict emissions of CO <sub>2</sub> , N <sub>2</sub> O and N <sub>2</sub> from agricultural soils	✓✓✓
ITE	United Kingdom	Grasslands, forest				✓
NCSOIL/NCS WAP	USA	Arable	Daily, less if infiltration	initial soil conditions, details of crops grown, temperature, rainfall, and crop management information	2 residue pools, 4 SOM pools (2 microbial biomass, 1 mineralisable N, and 1 stable OM)	✓✓✓
ROTHC	United Kingdom	Soils in various systems	Monthly	Requires FOM as input in 'forward' mode.	Shares many concepts with CENTURY. Can be used to calculate OM inputs in 'reverse' mode.	✓✓✓
SOMM	Russia	Forest	Daily	total litter mass (crop and root residues), nitrogen and ash contents of the litter, soil water status time series (daily or monthly), soil temperature, initial SOM and N contents in each of the three compartments	Describes humus formation. 3 pools: litter, decomposed litter, topsoil humus.	✓
Verberne/MO TOR	Netherlands	Grasslands		Not known	3 FOM pools (decomposable,	✓

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Hybrid	United Kingdom	Ecosystem	Daily	Daily solar radiation, min, max temperatures, rainfall. Initial tree data. Soil water release parameters, nutrient contents.	structural, resistant), 3 SOM pools (non-protected, protected, stabilised) SOM routines are the same as CENTURY	Not known
ICBM	Sweden	Arable	Daily	Not known	Simple two-compartment model. Does not include plant processes.	Not known

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## 4 Conclusions and future research priorities

### 4.1 Overview

Our analysis of published literature (Chapter 2), as well as our modelling exercises (Annex I) indicate that there is potential for soil carbon sequestration under bioenergy crops in the U.K. The principal factors that will determine the rate of change (gain or loss), as well as the total amount of soil carbon affected by SRC willow or miscanthus plantations are:

1. Carbon inputs to the system (i.e. net primary production)
2. Decomposition rates of each of the major soil carbon pools (effects of temperature and moisture status)
3. Initial carbon content of the soil (an inverse relationship with sequestration rates)
4. Crop management is another factor (e.g. harvesting and re-plantation intervals, re-plantation practice)
5. Depth of soil which is being influenced by the bioenergy crop (affects total amount of carbon sequestered)

Our model was developed using a simple mass balance approach to estimate carbon inputs and outputs in a bioenergy crop system on an annual time-scale. Furthermore, the decomposition rate constants for each of the soil C pools were derived from a single natural regeneration woodland site close to Rothamsted in southern England. Thus, the value of this site-specific model is as a tool to identify the major controls on soil carbon sequestration under bioenergy crops, and to determine if there is a *potential* for soil carbon sequestration under SRC at that locality.

In the following calculations, we present an extrapolation of the model results to a larger area within the U.K. It is an hypothetical exercise intended to give a broad indication of the possible relative contributions of the fossil fuel replacement bioenergy crop component and the soil carbon sequestration component to overall CO<sub>2</sub> emission mitigation targets. The extrapolation assumes that all plantations across the country would behave in an identical manner to the hypothetical site at Geescroft.

#### Specific assumptions:

SRC plantation area:	125,000 ha	
SRC yield:	15 tonnes DM ha <sup>-1</sup> y <sup>-1</sup>	6.0 Mg C ha <sup>-1</sup> y <sup>-1</sup>
Soil C sequestration rate:		0.5 Mg C ha <sup>-1</sup> y <sup>-1</sup>

#### CO<sub>2</sub> emissions mitigation:

Target net reduction (Section 1.1.1)		12.7 Tg C y <sup>-1</sup>
SRC harvested bioenergy crop:	$125,000 \times 6.0 \times 10^6 =$	0.75 Tg C y <sup>-1</sup>
(i.e. fossil fuel replacement component assuming a 1:1 replacement)		
SRC soil sequestration component:	$125,000 \times 0.5 \times 10^6 =$	0.0625 Tg C y <sup>-1</sup>

Thus, the model estimates that the total contribution of SRC plantations to mitigation of C emissions would be 0.8125 Tg C y<sup>-1</sup>, or about 6% of the target net reduction.

Actual quantification of soil carbon sequestration under bioenergy crops across the U.K. will require a much more detailed modelling approach that accounts for the dynamics of the system within each season, as well as for regional variations in yield and soil carbon inputs and outputs. In addition, a substantial amount of experimental data above- and below-ground is required for both willow and miscanthus plantations. Our review of current literature indicates that fundamental information on biomass and production rates for many of the major carbon pools in a bioenergy crop system are not available. In particular, we found almost no data on the distribution of carbon pools below-ground, or on their individual rates of change for bioenergy crops (or woodland in general) in the U.K. Thus, the goal of developing a general model to predict rates and quantities of soil carbon sequestration under bioenergy crops across the U.K. will require:

- a model that runs on a weekly or monthly time-scale to be responsive to local climatic conditions
- above- and below-ground biomass and production data
- sub-routines to estimate decomposition rate constants on the basis of local soil temperature and drainage conditions
- initial soil C contents
- sub-routines appropriate to SRC willow and miscanthus crops

In order to achieve this goal, we suggest the following technical recommendations for ongoing bioenergy crop trial studies in the U.K., as well as highlighting general priorities for future research.

## 4.2 Technical recommendations for soil carbon sequestration studies

### 4.2.1 Definitions and Methodology

Total carbon in temperate soils is predominantly in organic form, although some inorganic carbon may be present as carbonate minerals (Nelson and Sommers 1982). Soil organic carbon consists of the cells of micro-organisms, residues of plants, animals and micro-organisms at various stages of decomposition, and highly carbonised compounds such as charcoal and graphite (Nelson and Sommers 1982). The residues undergo a process of humification driven by chemical (rather than enzyme) reactions that result in dark-coloured organic material with increased concentrations of lignins and polyphenolics (Paul and Clark 1996). Because decomposition of humic pools is slow relative to fresh plant tissue carbon inputs, the size of the humic pools in soils are far greater than fresh pools. Soil carbon sequestration studies aim to quantify amounts of carbon sequestered below-ground that are significant from a national CO<sub>2</sub> emissions perspective. The quantities of carbon that can be sequestered on an annual basis are very small relative to the resident humic soil organic matter pool. Thus, soil carbon sequestration research necessitates the precise and accurate measurement of very small changes in the concentration of soil organic matter over time.

Organic carbon in soil may be measured by first removing any inorganic carbon present, and then combustion to CO<sub>2</sub> and subsequent quantification by gravimetric, titrimetric, volumetric, spectrophotometric, or gas chromatographic techniques (Nelson and Sommers 1982). Pre-treatment with a mixture of dilute H<sub>2</sub>SO<sub>4</sub> and FeSO<sub>4</sub> is used to remove inorganic carbon. Afterwards, various 'wet' combustion oxidation procedures involving K<sub>2</sub>Cr<sub>2</sub>O<sub>7</sub> (potassium dichromate) in a closed system flushed with a stream of CO<sub>2</sub>-free air have been frequently utilised. However, it is apparent that there are small but significant discrepancies between these methods. In particular, methods such as the widely used 'Walkey and Black' procedure that rely on heat of dilution or minimal heating, do not give complete oxidation of organic compounds in soil (Nelson and Sommers 1982). A period of 30 min heating at a temperature of 150°C is strongly recommended during the dichromate phase to achieve complete oxidation (Nelson and Sommers 1982). The U.K. Soil Survey and Land Resources Centre utilises a method (Kalembasa and Jenkinson 1973) that involves heating to 165°C. Precision in the procedure is +/- 0.2%; the recovery factor is 1.03 (P. Loveland, *pers. comm.*). Alternatively, 'dry' combustion by heating a soil catalyst mixture in a resistance or induction furnace with a stream of CO<sub>2</sub>-free air may be used provided that the inorganic carbon content is negligible. This latter method was used in one of the studies reported here (Grigal and Berguson 1998), without any reference to potential inorganic carbon contents. Clearly, very considerable care is required in the choice of method because of the high levels of accuracy and precision required for soil carbon sequestration studies.

In addition, particular care is required in collecting and preparing samples. Total carbon as defined above is likely to be extremely variable even at spatial scales of a centimetre or less. Several studies indicate that the spatial variance of many soil properties is minimised at a sampling interval of about 25 m (P. Loveland, *pers. comm.*). Soil organic matter is defined as those organic materials that accompany soil particles through a 2 mm sieve (Nelson and Sommers 1982). Thus, collected samples should first be sorted to remove visible roots and leaf litter, and then sieved prior to grinding. Grinding should be sufficient that the samples can pass through a sieve with openings 0.5 mm or less in diameter to reduce errors due to the presence of occasional fragments of carbonate minerals in a predominantly non-calcareous matrix (Nelson and Sommers 1982). Therefore, results of soil carbon analyses need to be reported with substantial detailed descriptions of the sampling, soil preparation and technical measurement procedures if they are going to be useful in monitoring changes in soil carbon that are extremely small relative to the total soil carbon pool.

### 4.2.2 Parameters on which detailed data is required

- Depth distributions of soil carbon concentrations and bulk densities. Sampling should be down to at least 1.5 m but at progressively greater intervals with depth.
- Root biomass and production data along a similar depth gradient.
- Production data for leaf litter and wood detritus.
- The immediate impact of coppicing on root production and turnover
- The impact of stump removal during plantation re-establishment.

## 4.3 Future research priorities

### 4.3.1 Hydrological interactions

Bioenergy crops are anticipated to have substantial water requirements (Stephens, Hess et al. 2001). Soil carbon contents are generally high in relatively wet and cool parts of the country because soil organic matter decomposition is inhibited. The overall net effect on soil carbon sequestration of relatively high initial soil carbon concentrations, as opposed to relatively low

decomposition rates requires further research. However, if bioenergy crop water requirements do lead to drying out of surface soil layers, then enhanced soil carbon losses may occur. These considerations indicate that there is likely to be a strong interaction between hydrology and soil decomposition rates under bioenergy crops. Thus, effects of hydrology will be an important component of any general model to predict patterns of soil carbon sequestration under bioenergy crops across the U.K.

#### 4.3.2 *Plantation replacement*

If, as anticipated, it is necessary to replace the entire plantation stock every 25 years or so in order to retain vigorous shoot regeneration after coppicing, then substantial further research is required into methods of re-planting. Disturbance of the soil due to uprooting of established rootstock is likely to enhance oxygen availability, and promote soil organic matter decomposition. The magnitude of this effect is difficult to predict, but could result in substantial losses from the soil carbon pools accumulated under the lifetime of the previous bioenergy plantation. Experimental tests are required.

#### 4.3.3 *Fertilisation requirements*

Sustainable and efficient management of plantations will require that the amount and timing of fertilisation is appropriate to crop demand. Increased soil carbon sequestration (as well as management practices such as the application of organic manures) may reduce plant available nutrient pools because of enhanced nutrient immobilisation by soil microbes. Further research is required into the effects of changes in soil carbon status on soil nutrient cycling.

#### 4.3.4 *Soil type*

The impact of the soil profile on soil carbon sequestration rates under bioenergy crops is unknown. Soils with a high fine silt or clay content may have relatively low decomposition rates due to greater water holding capacity, as well as the protection of soil organic matter particles from microbial attack through the formation of organo-mineral complexes (Anderson and Paul 1984). In addition, certain marginal lands such as wet grasslands can have very distinctive soil profiles with highly organic surface peats underlain by predominantly mineral soil horizons. In such cases, surface drying may result in net losses in soil carbon from the surface, but enhanced soil carbon sequestration at depth.

#### 4.3.5 *Climate change*

Our analysis of the impact of initial soil carbon content on potential soil carbon sequestration obviously has implications for the location of plantations across the U.K. The concept of a threshold value between net carbon gain and net loss for each climatic locality also indicates that there is an upper limit to the total amounts of carbon that could be sequestered. Essentially, the introduction of a bioenergy crop on soil that has previously been intensively cultivated and depleted of carbon, promotes a relatively rapid re-accumulation to equilibrium levels. Since climate is a major determinant of this soil carbon equilibrium, changes in climate that enhance decomposition, are likely to reduce the total soil carbon sequestered under bioenergy crops.

#### 4.3.6 *Innovative techniques*

Thus far, only basic measurement techniques have been used to determine if there have been any changes in soil carbon content in bioenergy crop plantations since their establishment. Since the initial soil carbon pool is so large, it is difficult to detect relatively small changes that might be nevertheless be important and significant from a CO<sub>2</sub> fossil fuel emissions mitigation perspective. Two recently developed methodologies that focus on determining carbon isotopes using isotope ratio mass spectrometry may be particularly useful in determining actual rates of soil carbon sequestration under bioenergy crops.

Miscanthus is a plant that photosynthesises using the C<sub>4</sub> biochemical pathway for carbon reduction (see Chapter 1). As a result, the carbon accumulated by miscanthus tends not to be strongly fractionated between the <sup>12</sup>C and <sup>13</sup>C carbon isotopes. Therefore, CO<sub>2</sub> respired during the decomposition of miscanthus tissue tends to have a substantial <sup>13</sup>C content. By contrast, most plants in temperate latitudes use the C<sub>3</sub> photosynthetic pathway which fractionates strongly against the <sup>13</sup>C isotope. Accordingly, CO<sub>2</sub> respired during the decomposition of non-miscanthus tissue tends to have a relatively low <sup>13</sup>C content. Thus, it is possible to quantify the contribution of miscanthus tissue to soil organic matter in a plantation by using a net carbon balance approach combined with measurements of the <sup>13</sup>C content of respired CO<sub>2</sub> from soil organic matter and a mixing model (Ehleringer, Buchmann et al. 2000), (Garten and Wullschleger 2000).

The second technique involves the measurement of <sup>14</sup>C isotopic ratios from soil organic matter CO<sub>2</sub> emissions (Richter, Markewitz et al. 1999). In the early 1960s, nuclear tests resulted in enhanced <sup>14</sup>CO<sub>2</sub> in the atmosphere which has since undergone radioactive decay, as well as being rapidly accumulated in plant biomass and in soil organic matter. Effectively, the proportion of <sup>14</sup>CO<sub>2</sub> being respired from soil organic matter may be used to determine the relative contributions of recent (post-1960s) and old (pre-1960s) carbon pools in soil organic matter. Therefore, if a bioenergy plantation has enhanced the sequestration of soil organic matter since the 1980s, the sizes of the 'recent' soil organic matter pools will be increased, and it may be possible to detect this change using the <sup>14</sup>CO<sub>2</sub> isotope ratio.

### **ANNEX I: Modelling carbon flows in biomass crops**

### **ANNEX II: Bibliography**

# **Review of the potential for soil carbon sequestration under bioenergy crops in the U.K.**

## **Annexes to Scientific Report**

**ANNEX I: Modelling carbon flows in biomass crops**

**ANNEX II: Bibliography**

**Paul Grogan  
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**MAFF report on contract NF0418**

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A preliminary desktop modelling study was carried out to predict the general time course of various carbon cycling processes, including C sequestration and bio-fuel production, over several short rotation coppice (SRC) cycles. Our analysis relies heavily on the dataset of long-term changes in soil organic carbon (C) at the Geescroft Wilderness site at Rothamsted, UK (Poulton 1996). This site was an area of land previously in arable cropping for several centuries that was fenced off in the 1880s and left unattended. Initially, the area was colonised by damp-loving grass species, but by 1957 it had reverted to woodland, and most grassland species had disappeared. The site is now an oak-dominated deciduous woodland with a few ground cover plants. Soil samples were taken in 1883, 1904, 1965, and 1985. The data show a steady increase in soil organic carbon in the 0-23 cm layer from 28 Mg C ha<sup>-1</sup> to over 60 Mg C ha<sup>-1</sup>, a mean annual increase of 0.33 Mg C ha<sup>-1</sup> y<sup>-1</sup>. During this time, the soil pH declined from 7.1 to 4.2.

Our approach has been to develop a model that can predict the tree growth and changes in soil carbon data from Geescroft Wilderness. Once calibrated in this way, we have modified key parameters to match tree production data and management practices associated with SRC (regular harvesting and plantation re-establishment cycles) to estimate plant and soil carbon pools expected in bio-energy plantations. A major assumption is that soil processes are similar between the two systems - carbon inputs to the soil may be different, but once the carbon is in the soil, its transformations are similar. Later, we do examine the implications of this assumption, particularly in relation to the initial sizes and decomposition rates of the carbon pools in different soils.

1.1 Description of the model

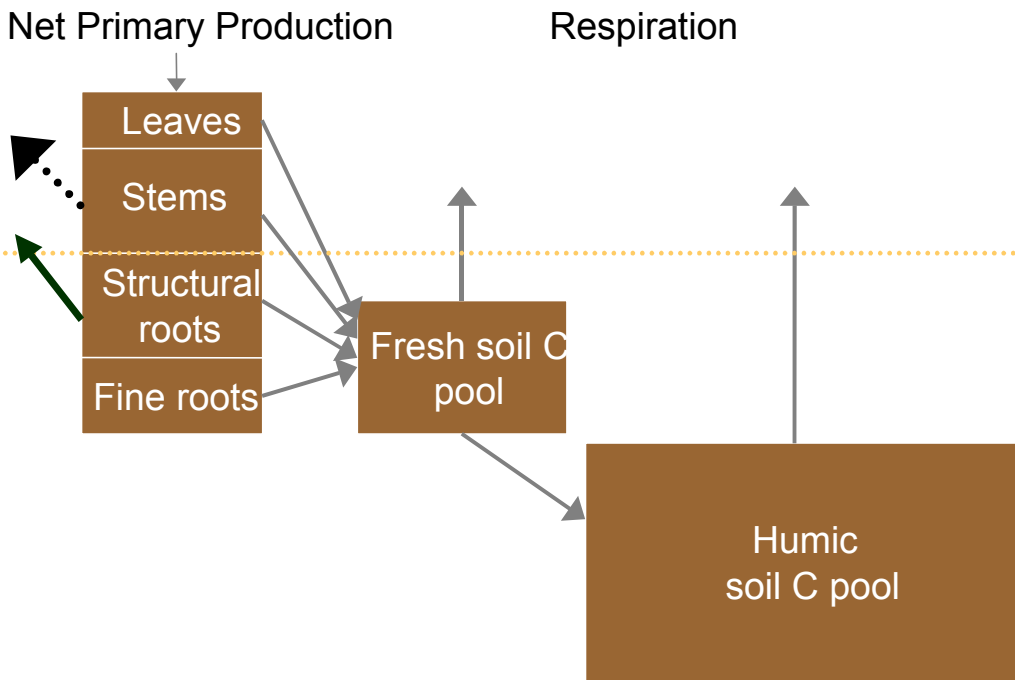


Figure 1. Schematic diagram showing the main carbon pools and fluxes used in the SRC model. The black lines on the left of the diagram indicate carbon losses due to crop harvests (dotted line) and plantation re-establishment (solid line).

A simple spreadsheet model was developed to simulate the main flows of carbon in stands of trees. No pretence is made that this is in any way superior to the models discussed in the previous section as it is a gross simplification of these, and does not include their level of detail. The purpose of this SRC model is purely as a tool to investigate the main carbon flows in biomass energy crops, and the potential contribution that such crops might make to offsetting carbon emissions to the atmosphere over the duration of a series of crop harvests and periodic plantation re-establishment events.

The SRC model runs on a yearly time-step, which is appropriate for investigating changes in soil carbon over a period of 100 years.

### 1.1.1 Biomass production

Annual biomass production was calculated using a version of Beer's Law:

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

where  $dW/dt$  is the growth rate of tree biomass (above- and below-ground) expressed in units of carbon ( $\text{kg C ha}^{-1} \text{y}^{-1}$ ),  $S$  is the annual receipt of short-wave solar radiation ( $\text{MJ m}^{-2} \text{d}^{-1}$ ),  $e_r$  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 ( $\text{m}^2 \text{leaf m}^{-2} \text{ground}$ ), and  $f_c$  is the fraction of carbon in the biomass ( $\text{g C (g DM)}^{-1}$ ). A factor of 10 is included to convert  $\text{g C m}^{-2}$  into  $\text{kg C ha}^{-1}$ . For both the woodland and SRC systems, values of  $k = 0.6$ , and  $f_c = 0.4 \text{ g C (g DM)}^{-1}$  were assumed. We recognise that  $f_c$  will vary slightly between different tissue types (i.e. leaf, stem, coarse roots and fine root material), but as yet there does not seem to be definitive figures available for willow coppice. However,  $0.4 \text{ g C (g DM)}^{-1}$  falls within the range of  $0.38 \text{ g C (g DM)}^{-1}$  for fine roots and  $0.46 \text{ g C (g DM)}^{-1}$  for stem wood given by Matthews *et al.* (1994), and for the sake of simplicity we have used this single value for all tissue types throughout this analysis. Leaf area index was calculated as described in below.

For the Geescroft Wilderness site, we found that a value for  $e_r$  of  $0.21 \text{ g MJ}^{-1}$  gave realistic behaviour that matched observed data. We recognise that this value subsumes many different processes (e.g. tree density, growth rates, seasonal variation in C assimilation, etc.), and moreover, is unlikely to have remained constant over the 100 years under consideration, particularly as regeneration appears to have been in two phases – woody shrubs and grass species until the 1950s, and full woodland since then (Poulton 1996). A more sophisticated model would take these distinct phases into account, but for the sake of simplicity, we have assumed a constant  $e_r$  representing an 'average' conversion efficiency over the period.

Of the total net carbon assimilated each year, it is assumed that a fixed proportion ( $f_R$ ) is allocated to new root growth, with the remainder (i.e.  $(1 - f_R)$ ) allocated to above-ground growth. A value of 0.25 was assumed for  $f_R$ , which is around the value given by Lambers *et al.* (1998).

Using these values, and assuming a closed canopy annual and solar radiation receipts of around  $4000 \text{ MJ m}^{-2} \text{y}^{-1}$  (commonly measured at Silsoe which is ~30 miles from Rothamsted), the model predicted an annual above-ground biomass production of almost  $6,300 \text{ kg DM ha}^{-1} \text{y}^{-1}$ , corresponding to about  $2,500 \text{ kg C ha}^{-1} \text{y}^{-1}$ . Net primary productivity (i.e. above- and below-ground growth) was predicted to be  $3350 \text{ kg C ha}^{-1} \text{y}^{-1}$ , closely matching the value of  $3300 \text{ kg C ha}^{-1} \text{y}^{-1}$  estimated by Jenkinson *et al.* (1992) for the site. About  $840 \text{ kg C ha}^{-1} \text{y}^{-1}$  was calculated by the model to be used for new root growth.

For the short-rotation coppice system, a value of  $e_r = 0.67 \text{ g MJ}^{-1}$  was chosen to match biomass production data measured on stands of short-rotation coppiced willow in the UK (Armstrong 2000). This gave annual wood yield figures of around  $14,600 \text{ kg DM ha}^{-1} \text{y}^{-1}$ . This higher value of  $e_r$ , compared to the regenerated woodland system reflects not only the faster growth rates per tree of short-rotation coppice, but also the higher planting densities associated with such systems.

### 1.1.2 Leaf area index

In the model, the maximum leaf area index ( $L$ ,  $\text{m}^2 \text{m}^{-2}$ ) achieved each year is one of the driving variables from which annual plant biomass production is calculated (see Equation 1), and is also used in the calculation of the amount of litter reaching the soil from the canopy (see Equation 2). As information is not available on the leaf area index in the Geescroft Wilderness dataset, for simplicity we have assumed that it increased linearly up to a maximum value of  $9 \text{ m}^2 \text{m}^{-2}$  after 10 years, and remained at that value thereafter. Such a pattern is realistic, as further leaf growth results in senescence of lower leaves due to shading, thereby maintaining  $L$  relatively constant despite the total above-ground biomass increasing every year. A value of  $9 \text{ m}^2 \text{m}^{-2}$  is within the range of  $L$  values for temperate deciduous forests of  $3\text{-}12 \text{ m}^2 \text{m}^{-2}$  given by Larcher (1995), and was consistent with the leaf litter production data for the Geescroft Wilderness site measured by Poulton (1996), as discussed in the next section.

For the short-rotation coppice system,  $L$  seems to be related to the age of the stand from time of planting. For willow planted at high density, Cannell *et al.* (1987) report values of maximum  $L$  of  $2.4 \text{ m}^2 \text{m}^{-2}$  in the first year after planting,  $4.5 \text{ m}^2 \text{m}^{-2}$  in the second year, and around  $7 \text{ m}^2 \text{m}^{-2}$  in the third year and thereafter. Interestingly, coppicing seems to have little effect on the maximum  $L$  reached in the following year (Persson and Lindroth 1994). We have expressed  $L$  in the short rotation coppice system, therefore, simply as a function of the age since planting ( $t$ , years) as  $L = \min(2.3t, 7)$ .

### 1.1.3 C inputs from canopy and root system

It is assumed that all leaf material that is produced in a given year falls at the end of the year, and enters the fresh carbon pool described in the next section. In addition, we assumed that there is an age-dependent amount of woody biomass that falls to the ground and which also enters the fresh carbon pool. This woody biomass includes not only branches and twigs that fall from the trees, but also trees that die due to self-thinning and other causes. We realise that this woody biomass has a slower decomposition rate than the leafy biomass, and that strictly it should be in a pool of its own. We consider the implications of this later in the Discussion, but we would argue that over the time period of 100 years, this assumption does not matter too much, and our purpose was to try and keep the model as simple as possible rather than simulate the system in detail.

The amount of C input into the fresh C pool represented by both of these inputs from the canopy ( $W_{Cin}$ , kg C ha<sup>-1</sup>) was therefore calculated as:

$$W_{Cin} = \frac{L f_C}{SLA} \times 10^5 + W_{AG} f_W \quad (2)$$

where  $SLA$  is the specific leaf area (cm<sup>2</sup> g<sup>-1</sup>), and  $f_W$  is the fraction of the above-ground weight of carbon ( $W_{AG}$ , kg C ha<sup>-1</sup>) that enters the fresh C pool as woody material rather than leaf litter. This fraction was calculated as  $f_W = 0.015/(1+100e^{-t})$  where  $t$  is the number of years since the start of the simulation in the case of the Geescroft Wilderness system, or since the last coppicing in the case of the SRC stands. Thus, early in the life of the woodland or the coppicing cycle, almost all of the input of C from the canopy is in the form of leaves, but as the stand grows older, progressively more woody biomass also enters the fresh C pool, up to a maximum of 1.5% per year of the above-ground biomass. In practice, in the short-rotation coppice system, however, very little woody material actually enters the soil, as most is removed through harvesting before it becomes old enough to fall. A value of 250 cm<sup>2</sup> g<sup>-1</sup> was assumed for  $SLA$ . For the Geescroft Wilderness woodland after 100 years, the model predicted about 3,000 kg C ha<sup>-1</sup> y<sup>-1</sup> entering the fresh C pool from the canopy (i.e. leaf + woody material), of which 1,440 kg C ha<sup>-1</sup> y<sup>-1</sup> was from leaf drop. The leaf drop value compares with that of 1570 kg C ha<sup>-1</sup> y<sup>-1</sup> measured by Poulton (1996).

It was assumed that a proportion ( $f_{FRTO}$ ) of the newly assimilated carbon allocated to the root system each year is lost as fine root turnover (i.e. root senescence) and enters the fresh C pool; this value also includes C lost from the roots through the processes of root respiration and rhizodeposition. In addition, we assumed that another fraction  $f_W$  of the plant carbon below-ground also enters the fresh C pool, representing death and decay of structural woody root biomass. This  $f_W$  is assumed to be the same as that calculated for the above-ground woody biomass. The total contribution to the C inputs into the fresh C pool from the root system ( $W_{Rin}$ , kg C ha<sup>-1</sup>) were therefore calculated as:

$$W_{Rin} = dW/dt f_R f_{FRTO} + W_{BG} f_W \quad (3)$$

where  $W_{BG}$  is the weight of carbon below-ground in the root system (kg C ha<sup>-1</sup>). For the Geescroft Wilderness site, we assumed that half of the below-ground carbon in the root system is lost to fine root turnover on an annual basis ( $f_{FRTO} = 0.5$ ), but that this value was higher in the short-rotation coppice system ( $f_{FRTO} = 0.85$ ) to represent the increased root senescence due to frequent above-ground harvesting.

In this way, the model predicted almost 750 kg C ha<sup>-1</sup> y<sup>-1</sup> entering the fresh C pool from the root system in the Geescroft Wilderness woodland after 100 years, with total annual C inputs from both the canopy and the root system together being 3,050 kg C ha<sup>-1</sup> y<sup>-1</sup>. As a comparison, in the evaluation of nine SOM models on long-term datasets (Smith, Smith *et al.* 1997), the models used the following values for C inputs into the system: RothC - 2850 kg C ha<sup>-1</sup> y<sup>-1</sup> (Coleman, Jenkinson *et al.* 1997); SOMM: 3140 kg C ha<sup>-1</sup> y<sup>-1</sup> (Chertov, Komarov *et al.* 1997); CANDY: 3000 kg C ha<sup>-1</sup> y<sup>-1</sup> (Franko, Komarov *et al.* 1997); NCSOIL: used an initial value of 2160 kg C ha<sup>-1</sup> y<sup>-1</sup> then a value of 4396 kg C ha<sup>-1</sup> y<sup>-1</sup> from 1957 onwards to simulate the conversion to full woodland (Molina, Crocker *et al.* 1997); while the Verberne/MOTOR model used a value of 3070 kg C ha<sup>-1</sup> y<sup>-1</sup>. Thus, our value of 3,050 kg C ha<sup>-1</sup> y<sup>-1</sup> falls well within the range of values used in these models.

#### 1.1.4 Decomposition of soil carbon pools

For simplicity, it was assumed that there were two organic carbon pools in the soil, that of fresh organic carbon originating from recent tree growth, either from the canopy or from the root system, and that of long-lived soil organic carbon (or humus). As in many of the models reviewed previously, decomposition of these two pools was assumed to follow first-order kinetics with different rate constants:

$$dC_p / dt = -C_p k_p \quad (4)$$

where  $dC_p/dt$  is the rate of change of the carbon pool  $p$  in question (kg C ha<sup>-1</sup> y<sup>-1</sup>),  $C_p$  is the size of the pool (kg C ha<sup>-1</sup>), and  $k_p$  is the rate constant (y<sup>-1</sup>).

Preliminary simulations indicated that obtaining realistic rates of decay of the two soil carbon pools were critical to the development of an accurate model. To do this, we used the Geescroft Wilderness data to calibrate the model by optimising for the two rate constants of each carbon pool,  $k_{FOM}$  and  $k_{HUM}$ . To facilitate the optimisation procedure, we maintained a constant ratio between the two rates (on a daily basis) of 500:1 (i.e. decomposition of the fresh organic carbon pool was 500 times faster than the humic carbon pool), based on the approximate ratio of the corresponding rates used by the CENTURY model (Parton, Woerner *et al.* 1994). In this way, optimising the two rate constants to give the best fit to the Geescroft Wilderness soil C data produced values of 0.786 y<sup>-1</sup> for  $k_{FOM}$  and 0.0031 y<sup>-1</sup> for  $k_{HUM}$ , corresponding to turnover times of around 1.27 and 325 years, respectively. Expressed on a daily basis for comparison with other models, these values are 0.0042 d<sup>-1</sup> and 8.5×10<sup>-6</sup> d<sup>-1</sup>. As a comparison, Parton *et al.* (1994) give values used in the CENTURY model of 0.01-0.05 d<sup>-1</sup> for surface and buried litter carbon, 0.00054 d<sup>-1</sup> for the slow soil C pool, and 19×10<sup>-6</sup> d<sup>-1</sup> for the passive soil C pool. Molina *et al.* (1997) give calculated values for Pool III in the NCSOIL model (corresponding approximately to our humic carbon pool) ranging from 7×10<sup>-6</sup> d<sup>-1</sup> to 100×10<sup>-6</sup> d<sup>-1</sup>

<sup>1</sup>. It should be remembered in comparing these values that site differences in temperature, soil water status, and the C/N ratio of the organic material decomposed will all affect the actual rates of decomposition. Thus, our values of the two rate constants are well within the range of previous estimates.

Each year, losses of C to the atmosphere as CO<sub>2</sub> are subtracted from the pool in question, and any remaining carbon in the fresh organic pool is assumed to enter the humic carbon pool.

### 1.1.5 Tree harvesting

In the SRC stand, harvesting at periodic intervals was simulated by removing all of the above-ground woody biomass from the system. Root material was assumed to remain in the soil at these harvests. At longer intervals (~25 years), plantation re-establishment was simulated by removing all biomass, both above- and below-ground. We did not attempt to model any increase in decomposition rates of the humic fraction due to aeration of the soil at this time.

## 1.2 Results

In our model, we assumed a soil depth of 0.5 m, and that 80% of the carbon in the humic pool occurs in the top 0.23 m soil layer (for comparison with the Geescroft Wilderness data), and the remaining 20% in the 0.23-0.50 m layer. We ran the model for 100 years in each case for comparison with the Geescroft Wilderness site.

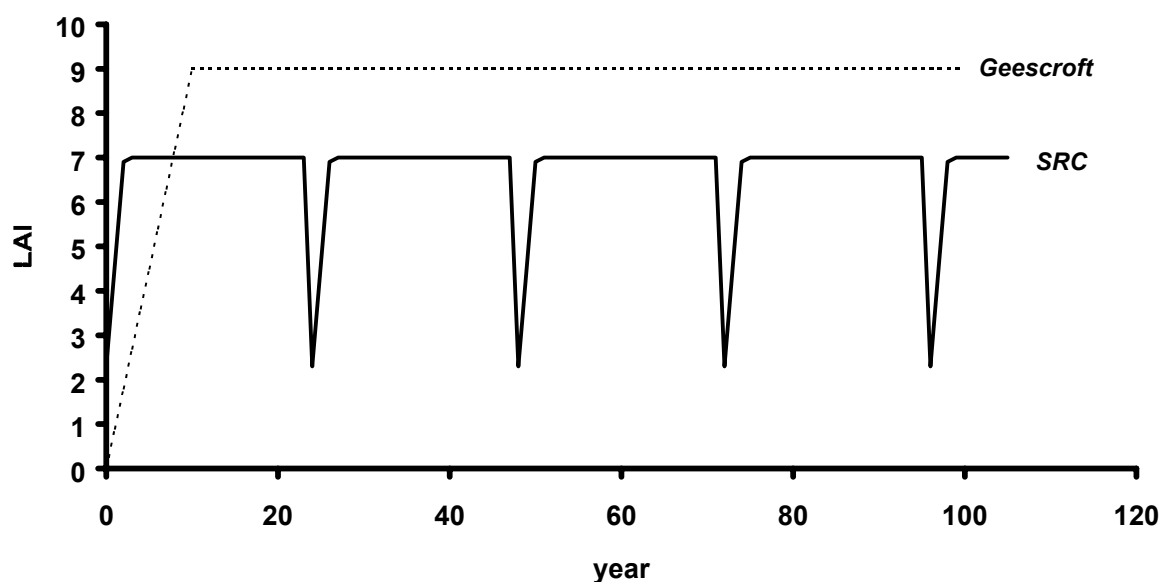


Figure 2: Predicted patterns of annual maximum leaf area index development for the Geescroft Wilderness woodland (dotted line) and for a short-rotation coppice stand (solid line) over 100 years. Harvest interval in the SRC is 3 years, and total removal for replanting occurs every 24 years.

### 1.2.1 Tree growth

Predicted patterns of leaf area index ( $L$ ) development for the two systems are shown in Figure . For the Geescroft Wilderness system, the maximum value of  $9 \text{ m}^2 \text{ m}^{-2}$  was reached after ten years and remained constant thereafter. It should be remembered that this is the maximum leaf area index reached during the year (i.e. in the summer), but that all leaves fall at the end of each year and fresh growth begins in the following spring. The predicted values probably overestimate  $L$  in the first part of the Geescroft Wilderness regeneration phase, when the vegetation cover was a mixture of woody shrubs and grassland. However, as there is no data available on the time-course of  $L$  at the site, we feel our approach is justified. In any case, errors in the estimation of  $L$  are likely to have only a small effect on biomass production when  $L$  is greater than about  $4 \text{ m}^2 \text{ m}^{-2}$ , as nearly all of the incoming solar radiation is being intercepted so that higher  $L$  values do not result in higher growth rates.

For the short-rotation coppice system,  $L$  rises and falls periodically as the stand is uprooted and replanted at regular intervals. The maximum value reached each year during each replanting cycle is  $7 \text{ m}^2 \text{ m}^{-2}$ , which agrees closely with values measured on stands of short-rotation willow at Silsoe (P. Martin, *pers. comm.*).

Predicted changes in above- and below-ground biomass for the two systems over 100 years are shown in Figure . The regenerated woodland rises steadily over the 100 years to reach a current value of about  $60 \text{ Mg C ha}^{-1}$  above-ground, and is still increasing. We were not able to find measured data for Geescroft Wilderness to verify this value, although Jenkinson *et al.*

(1992) did estimate that the current annual wood increment was about  $0.8 \text{ Mg C ha}^{-1} \text{ y}^{-1}$ . Assuming this was constant over 100 years (which is highly unlikely), this would give a current above-ground biomass of around  $80 \text{ Mg C ha}^{-1} \text{ y}^{-1}$ , somewhat higher than our value. However, this is almost certainly an overestimate, as the initial shrubby vegetation at the site for the first few decades would have stored much less carbon than the trees that are currently there, so our value of  $60 \text{ Mg C ha}^{-1}$  above-ground seems reasonable. In any case, the inputs of C into the soil are of more importance to the current exercise than is the amount of carbon in the standing biomass. The root C mass reaches a value of about  $20 \text{ Mg C ha}^{-1}$ , reflecting the partitioning ratio between above- and below-ground growth of 3:1.

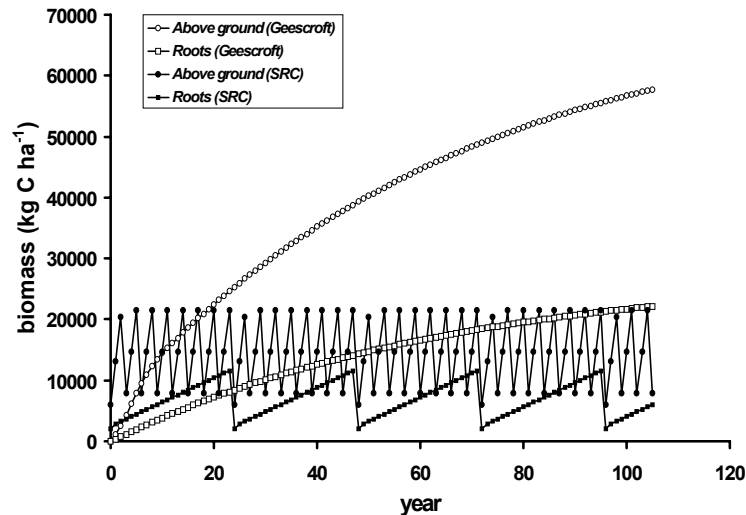


Figure 3: Predicted above- and below-ground biomasses ( $\text{kg C ha}^{-1}$ ) for the Geescroft Wilderness regenerated woodland and short-rotation coppice system over 100 years. Harvesting and replanting intervals as in Figure 2.

The above-ground biomass of the short-rotation coppice reaches about  $20 \text{ Mg C ha}^{-1}$  before being harvested – this corresponds to an average annual wood yield over the three-year coppice cycle of  $20/0.4/3 = 15 \text{ t DM ha}^{-1} \text{ y}^{-1}$ , yields typical of SRC stands in the UK (Armstrong 2000). Root biomass increases steadily until replanting, when it is removed completely.

### 1.2.2 Changes in soil carbon

Predicted changes in soil carbon for the Geescroft Wilderness site and a short rotation coppice stand assumed to be growing on the same site are shown in Figure 4. The essential differences between the two systems is the rate of C input (due to different  $e_r$  values), and the fact that the SRC system has biomass removed at periodic intervals. The average rate of sequestration of C in the soil of the Geescroft Wilderness system is  $0.40 \text{ Mg C ha}^{-1} \text{ y}^{-1}$ , while that in the SRC system is a little faster at  $0.51 \text{ Mg C ha}^{-1} \text{ y}^{-1}$ .

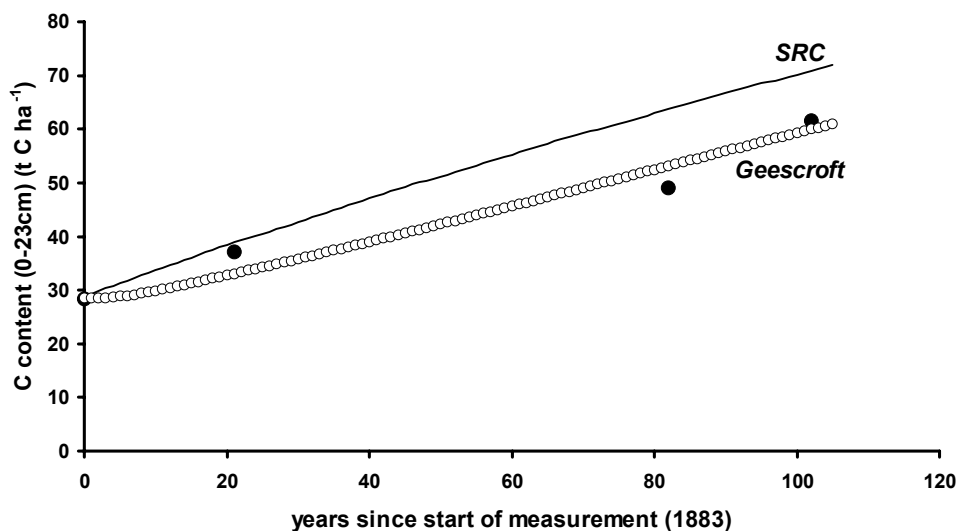


Figure 4: Changes in soil carbon in the 0-23 cm layer for the Geescroft Wilderness site (open circles) and for a short-rotation coppice stand (solid line) growing on the same site. Measured data points (filled circles) for the Geescroft Wilderness site are also shown. Harvesting and replanting intervals as in Figure .

### 1.2.3 Sensitivity analyses

Once the model was developed, we used it to identify the parameters to which soil C sequestration rates were the most sensitive. For this, we used a standard set of conditions, and varied the value of the parameter in question between a range of values, holding all the others constant. For the standard conditions, we assumed a short-rotation coppice growing on the same soil and in the same climatic conditions as at the Geescroft Wilderness site, with a harvesting interval of three years, and complete removal for replanting every 24 years (chosen to be a multiple of the harvest interval). For each simulation, we ran the model for 100 years, and used the annual rate of soil carbon sequestration averaged over the 100 years as the indicator variable to evaluate the sensitivity. Parameters were grouped into those that influenced the rate of C input into the system, and those that influenced the rate of loss of C from the system. We look at each of these in turn.

#### 1.2.3.1 Annual wood yield

The rate of C input to the fresh soil organic carbon pool was varied by changing the  $e_r$  value between the range of 0.2-1.6 MJ m<sup>-2</sup> d<sup>-1</sup>, encompassing the likely values for the regenerated woodland and the short-rotation coppice. This caused a range of wood yield values ranging from about 2 t DM ha<sup>-1</sup> y<sup>-1</sup> to nearly 45 t DM ha<sup>-1</sup> y<sup>-1</sup> (Figure 5). Over this range, and with all other parameters kept constant, the rate of soil C sequestration (averaged over the 100 years of the simulation) increased from around 0.2 Mg C ha<sup>-1</sup> y<sup>-1</sup> to 1.1 Mg C ha<sup>-1</sup> y<sup>-1</sup>. The slope of the line indicates that under these environmental conditions and harvesting regime, 22 kg C accumulates in the soil for every 1 tonne of wood yield. These figures suggest that the carbon sequestered in the soil in a short-rotation coppice plantation represents about 5.5% of the carbon accumulated as renewable bio-energy fuel.

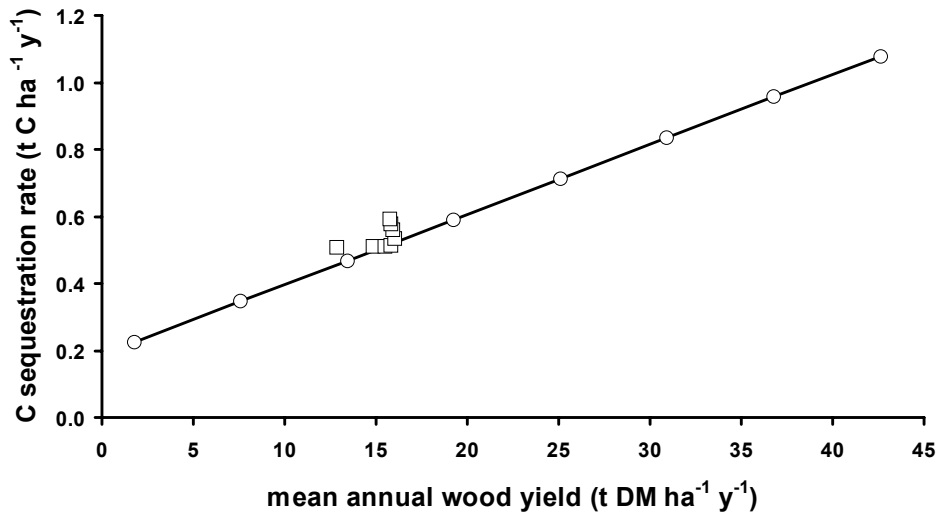


Figure 5: Relationship between mean annual wood yields (t DM ha<sup>-1</sup> y<sup>-1</sup>) and the rate of C sequestration in the soil (Mg C ha<sup>-1</sup> y<sup>-1</sup>) for a short-rotation coppice system (open circles). Variation in wood yields was obtained by varying the value of  $e_r$  between 0.2-1.6 MJ m<sup>-2</sup> d<sup>-1</sup>. Also shown are the same values for variations in harvest intervals described in Section 1.2.3.2 (open squares).

#### 1.2.3.2 Harvesting interval

Current practice is to harvest short-rotation coppice stands every 3-4 years. However, a management option open to growers is to vary the interval between harvests. The effect of different harvest intervals ranging from 1 to 10 years on average annual carbon sequestration rate is shown in Figure 6.

The model predicted that lengthening the interval between harvests slightly increases the rate of carbon sequestration in the soil. This increase is largely due to an increasing amount of woody biomass from both above- and below-ground that enters the soil C pools as the stand ages. For the same reason, average wood yield declines slightly after about 3-6 years, as an increasing amount of woody material is lost from the canopy. The relationship between average annual wood yields and soil C sequestration rates for the different harvest intervals is also shown in Figure 6. However, apart from very short intervals of less than 3 years, the predicted influence of harvest interval on yields is almost negligible.

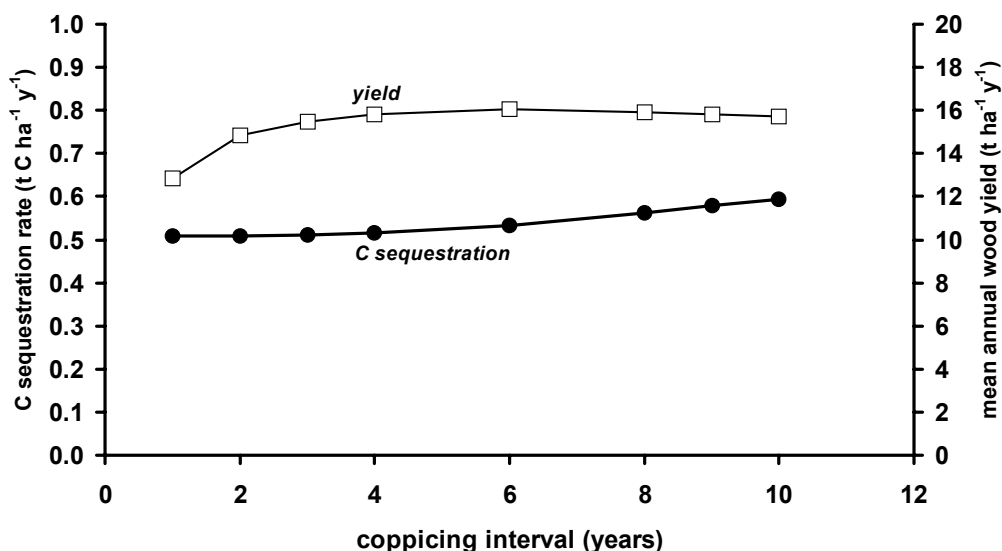


Figure 6: Predicted relationship between harvesting interval and mean annual soil carbon sequestration rate (filled circles) for a short-rotation coppice. Corresponding mean annual wood yields are also shown (open squares).

### 1.2.3.3 Total carbon inputs

Figure 5 showed the relationship between mean annual wood production and soil C sequestration rate. The relationship between wood production and mean annual soil C sequestration rate varies according to how this wood production was achieved – for example, the same level of wood production at different harvest intervals can give different soil C sequestration rates.

However, all of these can be brought into a common relationship by considering the total C inputs (i.e. from leaf, stem and root detritus) into the soil rather than the annual production (Figure 7). The slope of the line indicates that about 18% of the carbon entering the soil contributes to the increase each year, the rest returning to the atmosphere as CO<sub>2</sub> from microbial respiration. The y-intercept indicates that about 0.09 Mg C ha<sup>-1</sup> y<sup>-1</sup> is lost from this particular soil in the absence of any C inputs, which represents the annual decomposition of the humic C pool. If only 18% of the incoming C contributes to an increase in this pool, then nearly 0.09/0.18 = 0.5 Mg C ha<sup>-1</sup> y<sup>-1</sup> of fresh C must be added to balance this loss, represented on the graph as the x-intercept where net sequestration is zero. This value will, of course, depend on the initial soil carbon content – for a soil with 4% organic carbon content, around 2.5 Mg C ha<sup>-1</sup> y<sup>-1</sup> of fresh C must be added to balance the losses by humic decomposition.

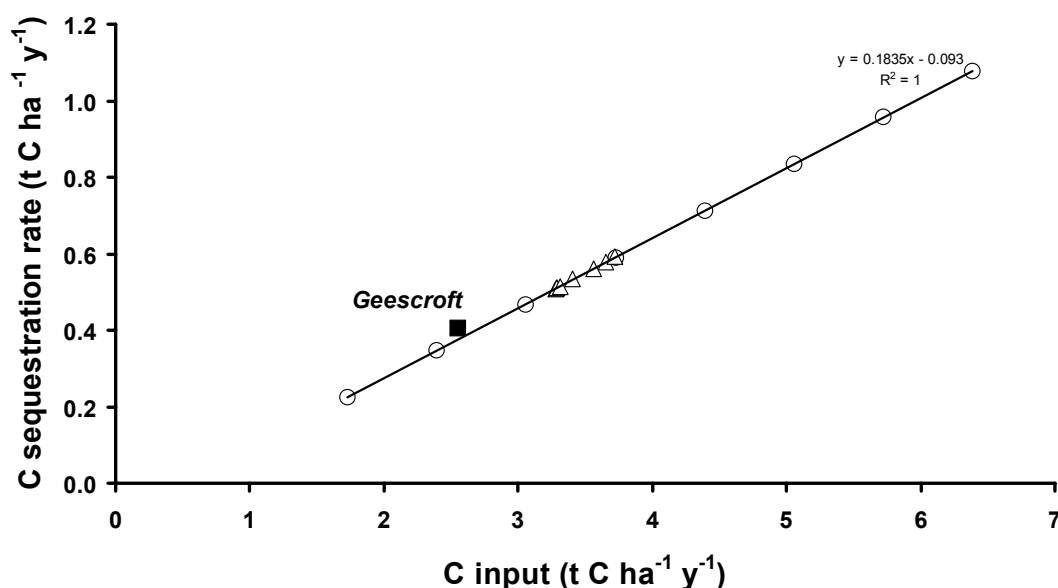


Figure 7: Relationship between total annual C inputs to soil (from above-ground and below-ground, Mg C ha<sup>-1</sup> y<sup>-1</sup>) and the rate of soil C sequestration (Mg C ha<sup>-1</sup> y<sup>-1</sup>) to a depth of 50 cm under a short-rotation coppice system.

### 1.2.3.4 Decay rates of soil carbon pools

Figure 8 shows the effect of changing the decomposition rates of the soil organic carbon pools. We have assumed that the ratio of the two rates, (i.e. those of the fresh carbon pool and the humic carbon pool), remains the same at 500:1 in each case. As expected, increased decomposition rates decrease the rate of soil C sequestration as more C is lost as CO<sub>2</sub> to the atmosphere. The rate of C sequestration is relatively sensitive to decomposition rates below about  $15 \times 10^{-6} \text{ d}^{-1}$ , but becomes less sensitive above this value. The value of  $19 \times 10^{-6} \text{ d}^{-1}$  given by Parton *et al.* (1994), for tropical ecosystems is shown for comparison, roughly twice as high as our estimated value.

As another comparison, we used the soil organic matter subroutine incorporated in the CERES family of crop models to estimate the mean decomposition rates of each pool over a year. This submodel is based on the PAPRAN submodel described by Seligman & van Keulen (1981). As in our model, two pools are assumed – a fresh organic matter (FOM) pool, and a more stable organic or humic pool (HUM). However, the FOM pool is further divided into three arbitrary pools corresponding approximately to the carbohydrate, cellulose, and lignin fractions. Each of the three FOM pools is assumed to have a different potential relative rate of decay – under non-limiting conditions, the decay constants ( $R_{p(max)}$ ,  $\text{d}^{-1}$ ) as reported by Seligman & van Keulen (1981) are 0.2, 0.05 and  $0.0095 \text{ d}^{-1}$  for carbohydrate, cellulose, and lignin respectively. However, these potential relative rates of decay are usually limited by soil temperature, soil moisture, and the C/N ratio of the decaying material, as described by Godwin & Jones (1991). A similar procedure is used to estimate the rate of decay of the humus pool ( $dO_H/dt$ ,  $\text{kg C ha}^{-1} \text{ d}^{-1}$ ), except that the potential relative rate of decomposition ( $R_{H(max)}$ ,  $\text{d}^{-1}$ ) is much slower, with a value of  $85 \times 10^{-6} \text{ d}^{-1}$ . The rate constants for the fresh carbon and humic carbon pools in our model were obtained by running the CERES organic matter sub-model with no crop present for one year using daily weather data for 1991-1997 from the Silsoe Campus weather station. This allowed the variation in temperature and rainfall throughout the year to be taken into account. In this way, mean values of  $0.0021 \text{ d}^{-1}$  and  $3 \times 10^{-6} \text{ d}^{-1}$  were obtained for the fresh carbon and humic carbon pools, respectively. These values are about half of the values we obtained for the Geescroft Wilderness data (Figure 8), and were low because the model predicted that for about four months of the year (i.e. November-February), decomposition did not occur at all because of low temperatures, and that for the remaining eight months, decomposition was slow (~20% of the potential rates) due to relatively low mean summer temperatures compared to the optimum of 30 °C. Whether this is the case in reality needs to be verified.

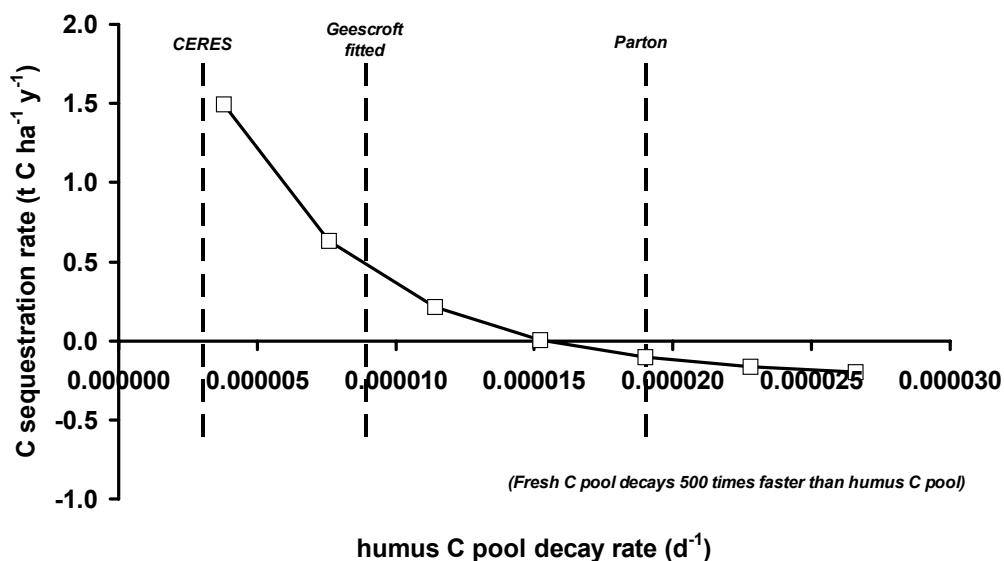


Figure 8: Relationship between decomposition rate of the humic carbon pool and the mean soil carbon sequestration rate to a depth of 50 cm under a short-rotation coppice system. The decomposition rate of the fresh carbon pool is assumed to be 500 times faster (on a daily basis) than that of the humic C pool in each case. The humic decomposition rate obtained by fitting the model to the Geescroft Wilderness soil data is shown, as are the values given by Parton (1994), and those calculated by the CERES model using Silsoe weather data.

Our values are about mid-way between the CERES estimates and the Parton *et al.* (1994) values, and are therefore not unreasonable. It is interesting to note that had we used the Parton *et al.* (1994) values, a small net decline in soil carbon at the Geescroft site would have been predicted, whereas if we had used the CERES values, soil C sequestration would have been about three times faster. However, it is clear that there is much uncertainty in the estimation of these decomposition rates, and spatial extrapolation to estimate the impact of SRC on soil carbon across the UK would require a model with specific decomposition rate subroutines that account for soil moisture and temperature conditions as well as soil organic matter C/N ratios.

### 1.2.3.5 Initial soil carbon content

The initial soil carbon of the soil into which short-rotation crops are to be established is also important in determining whether there is likely to be sequestration of soil carbon or not. The rate of loss of carbon from the humic pool is dependent both on the rate constant discussed above, and on the actual size of the pool. Thus, in soils with a high soil organic carbon (SOC) content, loss of carbon through humic decomposition will be larger than that for a soil with a lower SOC content, even if the rate constants are the same in both cases. A larger input of carbon from the trees will therefore be necessary to offset this greater loss.

Figure 9 shows the mean annual soil carbon sequestration rate for a range of initial SOC contents. For ease of comparison with the Geescroft Wilderness data, we have expressed SOC content in terms of that in the 0-0.23 m layer, assuming that 80% of the carbon to a depth of 0.5 m is in this 0-0.23 m layer. It can be seen that there is a linear decline in the sequestration rate as the size of the humus pool increases, with a net loss occurring at SOC levels above 6%. Ironically, the Geescroft Wilderness soil at ~1% SOC is almost at the bottom end of the range of naturally occurring SOC contents, where carbon sequestration rates would be expected to be maximum. Soils with higher SOC contents would be expected to sequester carbon at slower rates.

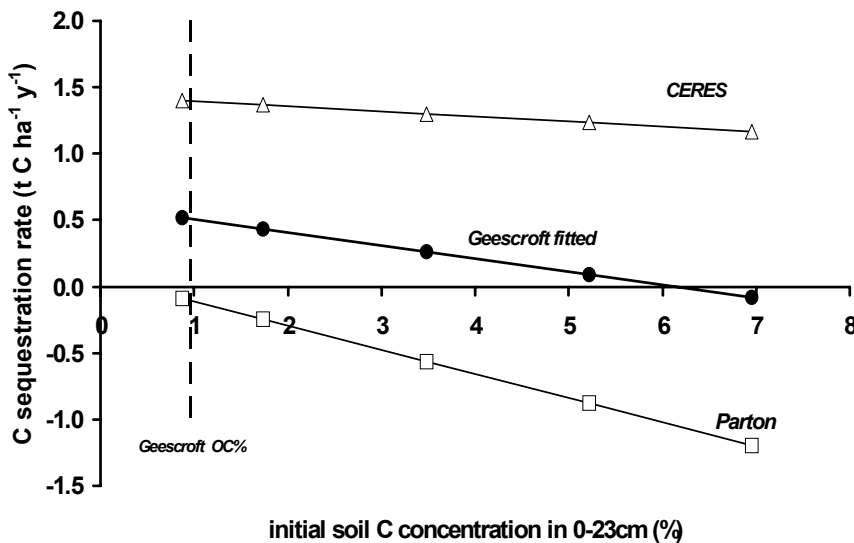


Figure 9: Relationship between initial soil organic carbon concentration in the 0-0.23 m layer and the mean soil carbon sequestration rate ( $\text{Mg C ha}^{-1} \text{y}^{-1}$ ) to a depth of 0.5 m under a short-rotation coppice system using decomposition rates obtained by fitting to the Geescroft Wilderness data (filled circles). Also shown is the corresponding relationship for decomposition rates given by Parton (1994) (open squares) and calculated by the CERES model using Silsoe weather data (open triangles). The Geescroft Wilderness soil organic carbon concentration in 1883 is shown for comparison.

Also shown on the same graph is the corresponding relationship between initial soil organic carbon content and the predicted carbon sequestration rate using the values for the humic carbon pool decomposition rates given by Parton *et al.* (1994), and also those calculated by the CERES model using Silsoe weather data. The Parton values, which are for tropical agroecosystems under higher temperatures and hence will have faster decomposition rates, indicate that there would be no carbon sequestration under a short-rotation coppice system at any level of initial soil organic carbon. This highlights the general difficulties in increasing organic matter levels in tropical soils. The CERES values, on the other hand, indicate that sequestration of carbon in the soil would be relatively rapid, and not all that sensitive to initial SOC content.

## 1.3 Discussion

Our approach to modelling soil C sequestration under SRC plantations has been relatively simple. For example, no distinction is made between 'leafy' and 'woody' carbon entering the litter layer from above-ground, or structural root and fine root carbon entering the soil from below-ground. Each of these will have different decomposition rates, but we have lumped them into the same C pool with some kind of average decomposition rate. This is probably the largest weakness of the current model, and we have not attempted within the time-frame of this review to explore the implications of the approach. However, our feeling is that it probably does not matter too much when we are dealing with time periods of the 100 years or so, but for shorter periods, it would definitely be necessary to introduce at least one other carbon pool with a decomposition rate intermediate between the two that we have used. Most of the SOM models reviewed in the previous section have at least three carbon pools, and some as many as six.

For simplicity, we have also assumed that the carbon that is decomposed from both pools is lost to the atmosphere as  $\text{CO}_2$ , and that any remaining C in the fresh carbon pool is transferred to the humic carbon pool at the end of the year. In reality, this remainder is probably largely decomposed at a faster rate than that of the humic pool. In addition, a fraction of the decomposed

C will enter a microbial C pool, and at a later stage enter the humic C pool. A more rigorous model would explicitly account for microbial carbon cycling activity, and incorporate the critical microbial linkages associated with changes in the nitrogen content of decomposing material.

We have also not considered the effect on decomposition rates of uprooting stumps for replanting every 25 years or so. This would result in considerable soil disturbance with greater aeration leading to higher oxygen penetration into the soil profile, and higher decomposition rates of the humic pool would be expected. These effects could last for some years after replanting, so that a proportion of the carbon sequestered in the previous planting cycle would be lost again as CO<sub>2</sub> to the atmosphere. Mean C sequestration rates over long time periods would, therefore, be lower than we have predicted.

Nevertheless, despite these simplifications, the model incorporates the essential processes underlying soil carbon dynamics, and therefore is appropriate to investigating basic questions relating to carbon sequestration under short-rotation coppice. Sensitivity analysis showed that the model was most sensitive to the efficiency of conversion of light into biomass ( $e_r$ ), the decomposition rate constants of the two soil carbon pools,  $k_{FOM}$  and  $k_{HUM}$ , and the initial size of the two pools (i.e. the initial carbon content of the soil). This is not surprising, as  $e_r$  controls the overall input of C into the system through assimilation into organic materials through photosynthesis, and  $k_{FOM}$ ,  $k_{HUM}$  and the initial size of the soil carbon pools control the rate at which C is lost back to the atmosphere as CO<sub>2</sub> through microbial respiration during decomposition.

The issue of the size of the initial humic carbon pool is particularly relevant. It has been proposed that land suitable for growing short-rotation coppice for bio-energy is on wetter soils, in Wetness Class III and IV (Jarvis and Mackney 1979). These would typically be under long-term grass, which tend to have a relatively high soil organic carbon content of around 4-5% in the surface horizon. If this is the case, the results shown in Figure 9 would suggest that carbon sequestration rates in such soils are likely to be minimal. However, it has been shown that the relationship between carbon sequestration rate and initial soil carbon content is highly dependent on the values of the humic carbon pool decomposition rates, which also have a large degree of uncertainty associated with them, with a six-fold difference between the values calculated by the CERES model and those given by Parton *et al.* (1994). As the Parton values are for tropical ecosystems, it is more likely that in UK conditions, the true values rates are closer to those predicted by the CERES model, but clearly there is a need to obtain more reliable estimates to be sure of this.

In the case of Wetness Class III and IV soils, decomposition rates may approach those predicted by the CERES model. However, net carbon balance depends on both inputs and outputs, and such low decomposition rates may not necessarily result in the associated high soil C sequestration rates suggested in Figure 9, as wet conditions could also reduce plant growth. The challenge for developing effective strategies to promote C soil sequestration under bio-energy crops will be to promote C inputs to the soil whilst simultaneously slowing organic matter decomposition.

Our analysis assumes that decomposition rate constants are independent of soil carbon concentrations. In reality, however, there is often a negative correlation between soil carbon concentrations in soils across the UK and the associated decomposition rates, provided that they have not been intensively managed for agriculture. For example, where decomposition rates are low due perhaps to low temperatures or high soil water content, soil carbon concentrations are often high. Conversely, soil carbon concentrations tend to be low where conditions favour rapid decomposition. Therefore, SRC plantations may not have much overall net impact on soil carbon sequestration in natural and semi-natural ecosystems where soil carbon concentrations are inherently low. By contrast, the greatest potential for soil carbon sequestration may be on sites previously used for agriculture, where initial soil carbon concentrations are low due to land management practices (e.g. intensive ploughing), rather than inherently high decomposition rates.

Much of the economic success of short-rotation coppice systems depends on adequate wood yields being obtained. The review of literature earlier in this report (Chapter 2) indicates a wide range of reported yields from different sites in the UK. Selection for high yielding clones of willow is currently in progress. Using a range of  $e_r$  values in our SRC model to simulate clones with different growth rates indicated that the increased yields from faster growing clones will also increase the rates of carbon sequestration due to greater inputs of carbon into the system (Figures 5,7). Although it seems that  $e_r$  has a value of around 0.67 g MJ<sup>-1</sup> for typical short-rotation willow coppice stands reported by Armstrong (2000), Cannell *et al.* (1987) give values of 1.4-1.6 g MJ<sup>-1</sup> for willow, indicating that there is considerable scope for improvement in this direction.

## 1.4 Conclusions

The very preliminary modelling study described above suggests that short-rotation coppice systems have the capacity to sequester substantial amounts of carbon, comparable to, or even greater than, an undisturbed naturally regenerating woodland site. The removal of biomass from SRC at periodic intervals for energy production appears to be more than balanced by the higher biomass production per unit area in such systems, due both to faster growth rates per tree, and the higher planting densities used. Thus, inputs of C into the soil are greater in the SRC compared to the regenerated woodland.

However, the rate of sequestration depends very much on the initial carbon content of the soil, as the rates of humus decomposition are directly proportional to the size of the humus pool (as well as the decay constant). Soils with a high soil organic carbon content (4-5%) may exhibit high decomposition rates resulting in substantial ongoing carbon losses, such that relatively high soil carbon inputs to the soil will be required before any sequestration can occur. The model suggests that SRC plantations on such soils may result in a net loss of soil carbon.

Our study has shown that there is a severe shortage of reliable data in the area of carbon sequestration under biomass energy crops, as evidenced by the large number of assumptions that we have had to make in the development of the model. Future research should focus on obtaining more reliable estimates of the important parameters of such systems (see Chapter 4).

### 1.1 Useful websites

<http://www.detr.gov.uk/environment/climatechange/draft/section6/07.htm>

[http://www.etsu.com/en\\_env/](http://www.etsu.com/en_env/)

<http://www.dti.gov.uk/renewable/index.html>

<http://www.joanneum.ac.at/iea-bioenergy-task38/>

[http://www.fe.doe.gov/coal\\_power/sequestration/index.shtml](http://www.fe.doe.gov/coal_power/sequestration/index.shtml)

<http://bioenergy.ornl.gov/bfdpmain.html>

<http://www.esf.edu/willow/default.htm>

[http://www.esd.ornl.gov/research/tolbert\\_highlight.html](http://www.esd.ornl.gov/research/tolbert_highlight.html)

<http://www.ars.usda.gov/is/pr/2001/010216.htm>

<http://www.ars.usda.gov/is/AR/archive/feb01/bank0201.htm>

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