Short title: Soil carbon loss in bioenergy systems

Type?

Does soil carbon loss in biomass production systems negate the greenhouse benefits of bioenergy?

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Abstract

Interest in bioenergy is growing across the Western world in response to mounting concerns about climate change. There is a risk of depletion of soil carbon stocks in biomass production systems, because a higher proportion of the organic matter and nutrients are removed from the site, compared with conventional agricultural and forestry systems. This paper reviews the factors that influence soil carbon dynamics in bioenergy systems, and utilises the model FullCAM to investigate the likely magnitude of soil carbon change where bioenergy systems replace conventional land uses. Environmental and management factors govern the magnitude and direction of change. Soil C losses are most likely where soil C is initially high, such as where improved pasture is converted to biomass production. Bioenergy systems are likely to enhance soil C where these replace conventional cropping, as intensively cropped soils are generally depleted in soil C. Measures that enhance soil C include maintenance of productivity through application of fertilisers, inclusion of legumes, and retention of nutrient-rich foliage on-site.

Modelling results demonstrate that loss of soil carbon in bioenergy systems is associated with declines in the resistant plant matter and humified soil C pools. However, published experimental data and modelling results indicate that total soil C loss in bioenergy systems is generally small. Thus, although there may be some decline in soil carbon associated with biomass production, this is negligible in comparison with the contribution of bioenergy systems towards greenhouse mitigation through avoided fossil fuel emissions.

Key Words: bioenergy system, greenhouse gas balance, land use change, soil carbon

Introduction

Interest in bioenergy is growing across the Western world in response to mounting concerns about climate change due, in large part, to use of fossil fuels for energy. Biomass for conversion to bioenergy may be derived from waste sources such as sawmill residues and construction and demolition waste, where bioenergy is a desirable alternative to disposal to landfill. Biomass is also obtained from forestry and agriculture, either from dedicated systems, where the entire crop or forest is harvested for bioenergy, or as a by-product of crop or timber production. In most conventional crop and timber production systems a significant fraction of the biomass remains on site after harvest, whereas, bioenergy systems often remove the majority of above ground biomass at harvest. For example, wheat straw that would normally be plowed into the soil may be baled and removed after the grain has been harvested; branches that would usually decompose on the forest floor after timber harvest, can be compressed into bundles for efficient transport to a power plant. There is some concern that bioenergy systems may suffer from declining soil carbon. This paper reviews the factors that influence soil carbon dynamics in bioenergy systems, and, through modelling, investigates the likely magnitude of soil carbon change where bioenergy systems replace conventional land uses, and the impact on greenhouse mitigation benefits of bioenergy systems.

1. The link between soil carbon and bioenergy

The soil organic carbon pool, derived principally from plant matter, constitutes a fundamental dynamic link in the carbon cycle, and contains 50-80% of the carbon in a mature forest and over 95% of carbon in a crop or pasture system (Bolin et al., 2000).

Soil carbon stock at any one time reflects the balance between the inputs from plant residues and other organic matter, and losses due to decomposition, erosion and leaching. Thus, removal from the field or forest of a higher proportion of biomass in bioenergy systems, compared with crop or timber production systems, reduces the addition of organic matter to the soil carbon pool, and, therefore, may result in a decline in soil carbon stock. Loss in soil carbon may reduce long term productivity, and should be accounted in assessing the net greenhouse gas balance of bioenergy systems. The following sections consider the natural and anthropogenic factors that influence soil carbon stocks.

2. Natural processes affect soil carbon

Soil organic carbon comprises residues of plants, animals and microorganisms in various stages of decomposition. The major sources are turnover of leaf litter and fine roots. Factors that enhance plant growth, such as warm, moist climate and high soil fertility promote organic matter addition to the soil carbon pool.

Organic matter is decomposed through comminution and mineralisation, which are dependent on appropriate soil fauna and microorganisms (Spain and Hutson 1983; Cogle et al. 1995). Mineralisation, in which soil microbes digest organic matter, respiring C to the atmosphere and simultaneously releasing plant nutrients, is most rapid in warm, moist climates that promote microbial activity. Conversely, mineralisation is slow in situations where microbial activity is inhibited: low temperatures, limited moisture, low pH or limited oxygen due to waterlogging or soil compaction. Mineralisation rate is influenced by decomposability or "quality" of the organic residues, which is determined by chemical composition (Martens, 2000), being lowest for material low in nitrogen or high in recalcitrant fractions such as lignin and waxes (Oades, 1988; Van Cleeve and Powers 1995).

Cycling of carbon is inherently linked with cycling of nutrients, particularly nitrogen, because nutrients are returned to the soil as organic matter decomposes. In many environments nitrogen availability is the factor limiting plant productivity and, therefore, carbon inputs to the soil.

The influence of environment on rate of plant growth and turnover, and rate of mineralisation of soil carbon, leads to high soil carbon stocks in cool temperate forests and wetlands where plant productivity is relatively high but activity of soil biota is inhibited (Bunnell et al*.* 1977, Spain et al*.* 1983, Attiwill and Leeper, 1987; Bolin et al., 2000; Van Cleeve and Powers, 1995). Soil carbon stocks are lower in the wet tropics where organic matter is turned over rapidly, and lowest in dry environments where plant growth is limited (Bolin et al., 2000).

Soils with high active clay content, high base status, and those high in aluminium and iron, provide strong physical and chemical protection to organic colloids, through formation of intra-aggregate or organomineral complexes (Islam and Weil, 2000; Shepherd et al., 2001). Sandy soils have low capacity to stabilise soil carbon (Zinn et al. 2002). Thus, soil carbon stocks are high in soils with high active clay content (e.g. Andisols, Vertisols, Mollisols), followed by low activity clay soils (Oxisols, Ultisols) and low in all sandy soils (Scott et al., 2002; Nabuurs et al. 2003).

The soil organic carbon pool comprises components with different turnover times. The most rapidly decomposing, and often smallest, pool is the labile pool (microbial biomass, soluble C, light fraction and macroorganic matter), with a turnover time of 1-5 years (Parton et al., 1987). The least active pool is essentially inert organic matter, highly resistant to decomposition due to physical and /or chemical protection, that decays on a time frame of thousands of years, while the remaining soil carbon is contained in the humified pool, with turnover time of decades (Parton et al., 1987).

3. Land use affects soil carbon

In an undisturbed natural ecosystem, the additions and losses of soil organic carbon are balanced over time, and soil carbon stock reaches a stable equilibrium value. Similarly, in a managed forestry or agricultural system where land use practices have remained constant over long periods, inputs and losses will be in equilibrium, but if land use or management practices are altered so that the balance between inputs and decomposition is affected, soil carbon stock will change.

It is well known that intensively cropped soils have low organic matter content. This is often attributed to enhanced mineralisation due to cultivation (e.g. Mann, 1986; Attiwill and Leeper, 1987; Blair et al., 1995; Grigal and Berguson, 1998; Dalal and Chan, 2001), which aerates the soil, redistributes and exposes physically protected organic matter, encouraging microbial breakdown (Shepherd et al., 2001). However, there is evidence that soil disturbance *per se* has little impact on soil organic matter content (Paul et al., 2002). Probably of greater significance, cropping systems commonly suffer soil erosion, and include regular periods of minimal organic matter input during fallow and in early establishment (Dalal and Chan, 2001). Thus, cropping decreases soil carbon through a combination of reduced input and enhanced loss. The loss in soil C resulting from cultivation is greatest in soils initially high in soil C (Mann, 1986).

A change in land use from forest or grassland to cropping will, therefore, generally lead to loss of soil carbon (e.g. Whitbread et al., 1998; Murty et al., 2002). Guo and Gifford (2002), in their comprehensive review, found that soil C was reduced 50% or more when pasture land was converted to cropping. The loss of soil C was greatest (-78%) in areas with 400-500 mm precipitation, and peaked at 30 - 50 years after conversion (85% loss).

Conversely, conversion of cropland to pasture or forest is likely to increase soil carbon though it may decline initially (Smith et al., 1997; Romanya et al., 2000; Vesterdal et al., 2002). In the tropics, where soil C turnover is rapid, Binkley and Resh (1999) observed no impact on total soil C of conversion of sugar cane to Eucalyptus plantation after 32 months, however there was a sharp decline in cane-derived C, matched by increasing inputs from the plantation. Guo and Gifford (2002) concluded that, on average, replacing cropping with pasture or forest leads to an increase in soil C of 18-20%.

The impact on soil carbon of conversion from pasture to forest is less clear: some studies report a substantial decline (Gifford and Barrett, 1999; Scott et al., 1999; Post and Kwon, 2000; Turner and Lambert, 2000), though recent reviews have concluded that large losses are unlikely in most situations (Guo and Gifford, 2002; Paul et al., 2002). Soil C is likely to decrease initially, as a result of a decline in pasture litter inputs in the early phase of plantation establishment, and then increase as litter input from the forest is added to the system (Johnson, 1992; Smethurst and Nambiar, 1995; Grigal and Berguson, 1998). Significant losses of soil carbon after reforestation are most likely in soils that are high in labile carbon, such as where new plantations are established into pastures that have been heavily fertilised, and enhanced productivity has elevated the soil C above native levels.

As the plantation grows, soil carbon is replenished from litter fall and root turnover, usually restoring soil C to original stock within 30 years (Paul et al., 2002).

However, there is some evidence from chronosequence and paired-site studies that equilibrium soil carbon stock may be lower in plantations than in the pasture they replace (e.g. Gifford and Barrett, 1999; Turner and Lambert, 2000). Interpretation of results from such studies is complicated by confounding impacts of site factors, initial carbon content, and management practices applied to initial and subsequent land use. Between studies, comparisons are complicated by differences in depth sampled, sample preparation, analytical method, correction for bulk density change, and pools included (see Smith et al. 1998; Gifford and Barrett, 1999; Guo and Gifford, 2002; Murty et al., 2002; Paul et al., 2002). In their comprehensive reviews of published data, in which these limitations were recognised and minimised, Paul et al. (2002) and Guo and Gifford (2002) concluded that reforestation with broadleaf tree species does not change, or increases, soil carbon, while reforestation with pine species generally leads to a small decline in soil carbon stock of around 15%. However, this conclusion is based on limited data, and should be verified.

Possible explanations for lower soil C under plantations (particularly pines) compared with pastures are that trees provide less input of C to the soil due to:

- lower proportion of fixed carbon allocated to the roots (Lugo and Brown, 1993; Kuzyakov and Domanski, 2000))
- slower turnover of roots (Brown and Lugo, 1990; Johnson, 1992; Post and Kwon, 2000)
- slower decomposition of litter due to lower "quality" (Saggar et al., 2001)

- micrometeorological impacts: slower decomposition of surface litter because soil surface is generally cooler and drier under forest (Brown and Lugo, 1990; Grove et al., 2001), due to higher transpiration rates and shading.
- limited interaction of surface litter with soil, so that decomposition occurs at the soil surface (Oades, 1988)

The effect may be indirect, resulting from a decline in productivity due to:

- soil acidification under pines due to organic acids released from decomposing foliage (Motavalli et al., 1995)
- net loss of nutrients from the site due to asynchrony of nutrient supply and demand: nutrient supply is greatest during early growth while slash from previous rotation mineralises, when tree requirements are small and root exploration is limited, so nutrients, especially nitrogen, may be leached (Parfitt et al., 2003; Heilman and Norby, 1998); when required during the rapid growth phase prior to canopy closure, the nutrients are unavailable.
- Lower nitrogen inputs due to loss of N-fixing pasture legumes.

There may be enhanced loss of soil C due to the ability of the ectomycorrhizal fungi associated with *Pinus radiata* to oxidise organic matter inaccessible to grassland fungi (Chapela et al., 2001).

Knowledge of the factors influencing soil carbon allows prediction of the probable soil carbon dynamics under bioenergy systems.

4. Impacts of bioenergy systems on soil carbon

4.1 Bioenergy systems based on increased removal of biomass, existing land use maintained

Conversion from conventional grain or timber production to bioenergy systems based on increased removal of biomass may reduce soil carbon as a result of three impacts:

1. Increased removal of biomass will directly reduce organic matter additions to the soil. In long rotation forestry systems the biomass removed at harvest is relatively minor compared with the organic matter returned to the soil over the rotation, but in short rotation forests and bioenergy crops a greater proportion of total biomass produced is removed, so there is greater potential for soil carbon depletion.

2. Removal of plant biomass exports nutrients from the site, which will lead to loss of productivity in the long term unless these nutrients are replaced. Bioenergy systems that remove foliage and bark, particularly high in N, P, K and Ca, are at high risk of nutrient depletion compared with grain and timber production systems. Nitrogen deficiency, and soil acidification resulting from export of base cations (Olsson et al., 1996) are particularly likely. In the long term, these impacts on soil chemical fertility may limit plant productivity leading to reduced input to the soil carbon pool.

3. As soil carbon begins to decline, this will directly impact on plant productivity, due to the role played by soil organic matter in maintaining biological fertility (nutrient cycling)), chemical fertility (nutrient availability, nutrient retention) and physical fertility (aggregate stability, porosity, moisture retention) (Tisdall and Oades, 1982 ; Sherwood and Uphoff, 2000). Declining plant productivity will, in turn, further reduce soil carbon.

“Harvest index”, that is, the fraction of above ground biomass removed at harvest, is higher for bioenergy than conventional crops and forests. However, the impact of biomass removal on soil C will be determined not by harvest index but by the fraction of total biomass production that is removed. In forest systems, a considerable fraction of the net primary production (NPP) is directed to fine roots with fast turnover (Vogt, 1991; Janssens et al, 2002), rapidly entering the soil C pool; a further significant proportion of NPP is retained on site in litter fall and root biomass (Heilman and Norby, 1998). Similarly, in perennial grass systems a high proportion (up to 50%) of total biomass production remains on-site after harvest as litter and root biomass (Hansen et al., 2004).

While these predicted impacts of bioenergy systems are anticipated from knowledge of carbon and nutrient cycling, there is limited experimental evidence from which to verify long term impact of bioenergy systems on plant productivity and soil carbon. Some evidence of likely impacts is seen from studies of whole tree harvesting (WTH, i.e. removal of stem+crown). In New Zealand, growth and nutrition of second rotation radiata pine did not differ between WTH and sawlog harvest treatments; WTH combined with removal of forest floor reduced growth and foliar N only on a sandy infertile soil (Smith et al., 2000a), indicating that ecosystem C stock, including soil C, may be little affected by WTH. Similarly, Johnson et al. (2002) observed little impact of WTH on soil carbon stock in US forests, compared with sawlog harvesting, though at one of four sites the sawlog treatment had 17% higher regeneration biomass than the WTH treatment 16 years after harvest; this differential effect on plant growth is likely to lead to reduced soil C in the longer term. In their meta-analysis, Johnson and Curtis (2001) concluded that WTH reduces soil C by 6% on average, while sawlog harvesting increases soil C by 18%.

In a study on the impact of harvest residue management conducted on two soil types, Mendham et al. (2003) observed that residue retention increased exchangeable cations but had little impact on soil C pools; residue retention improved plantation growth only on the lower fertility soil. In an agricultural system, removal of hay 3-4 times per year, despite higher rates of inorganic fertiliser, led to lower soil carbon in comparison with periodic grazing (Franzluebbers et al., 2000).

Thus there is some experimental evidence of decline in productivity and/or soil C where all above ground biomass is removed, particularly in infertile soils.

In many conventional forest and crop management systems, residues are burnt after harvest. Conversion of such systems to bioenergy systems would therefore remove biomass that is otherwise lost through burning. Therefore, biomass removal will not change the inputs of carbon, though non-volatile nutrients that would otherwise remain in ash will be removed. In systems where residues are not burned, unless they are incorporated by tillage it is likely that much of the above-ground litter, especially coarse material, decays at the surface, rather than entering the soil carbon pool, particularly in systems with low soil faunal activity. Again, biomass removal will have limited impact on organic input to the soil.

Thus, increased removal of biomass in bioenergy systems is likely to cause some loss in soil carbon, but because a significant proportion of *total* biomass production is retained, including the root and leaf litter biomass that constitutes the major input to soil C, the impact on soil C of removal of biomass for bioenergy should generally be small.

4.2 Bioenergy systems entailing land use change

Besides increased intensity of biomass removal from current land use, alternative means of increasing biomass production for bioenergy include:

1. conversion of cropland to short rotation bioenergy crops, such as rhizomatous perennial grasses (eg Miscanthus, switchgrass *Panicum virgatum*) or short rotation woody crops (eg willow, poplar) that are mown/coppiced every few years and allowed to regrow from roots/stump, and replanted after several harvests;
2. conversion of cropland to long rotation forest plantations for timber plus biomass
3. conversion of pasture to short rotation bioenergy crops
4. conversion of pasture to long rotation forest plantation for timber plus biomass

It is likely that conversion of cropland (options 1 and 2) will significantly increase soil carbon, due to enhanced input from leaf litter and root turnover and less frequent fallow, compared with conventional cropping. For instance, Tolbert et al. (2002) observed increased soil carbon stock under switchgrass and sweetgum within 3 years of conversion from conventional agriculture; 16 years after conversion from cropping, miscanthus increased soil C by 14 t ha^{-1}, representing 29% of the C input from miscanthus over the 16 year period (Hansen et al., 2004). Similarly, though soil carbon in 7-8 year old hybrid poplar plantations did not differ from adjacent row crops, simple modelling led the authors to predict a 10-25 t ha^{-1} soil C increase over a 10-15 year rotation (Grigal and Berguson, 1998). Conventional wisdom suggests that soil C may accumulate to a greater extent under the "less intensive" management of Option 2, compared with Option1, however, data are not available to test this assumption. The reduced frequency of harvest under Option 2 may lead to greater C accumulation than Option 1; however, a higher stocking rate and probably higher fertiliser rate under Option 1 may lead to higher plant inputs, that will, at least partially, overcome the impact on soil C of more frequent harvest.

Options (3) and (4) may lead to small losses or gains, depending on the relative balance of organic inputs and decomposition rate under the old and new land uses (see Sections 2 and 3.)

Changes in the short term may be subtle, indicative of significant longer term impacts on soil C, not yet evident from measurement of total soil C. For example, Ma et al. (2000a) observed no change in soil C concentration over 2 years after conversion from pasture to switchgrass, however C turnover and microbial biomass C increased over the period. Similarly, Garten and Wullschleger (1999) observed no increase in soil C stock of switchgrass plots in comparison with tall fescue, five years after establishment of the switchgrass, although root carbon was at least 100% higher in the switchgrass. Where switchgrass replaced cropping, no change in soil C was detected over 2-3 yrs, though soil C was 45% higher at 0-15 cm after 10 years under switchgrass compared with the cropped/fallowed control (Ma et al. 2000b).

5. Land Management to Enhance soil carbon

Soil carbon is promoted by management practices that maximise organic matter inputs and minimise losses.

Residue management

Retain slash/crop residues rather than burning Retention of residues, rather than burning, increases organic matter input and protects against erosion of the carbon-rich surface soil, increasing soil C (Rasmussen and Parton, 1994; Ayanaba et al., 1976). In particular, foliage, bark and fine branches should be retained on site, as these are high in nutrients which also leads to high ash content, undesirable for most bioenergy applications. Retaining leaves is easiest to achieve with deciduous tree species, or perennial grasses with a dormant phase; for broadleaf species this can be achieved by windrowing or stacking branches in the field until the leaves drop, though this may pose an unacceptable fire risk in some climates.

Fertiliser

Apply fertiliser to overcome nutrient deficiencies and maintain fertility Fertiliser application, where it increases plant growth and therefore litter inputs, leads to soil C accumulation in forests (Johnson, 1992; O'Connell and Grove, 1993; Schroeder, 1991 and Turner and Lambert, 1986) and crops (Dalal and Chan 2001). Fertiliser rates and timing should be matched to the requirements of the crop/forest to maximise efficiency of fertiliser use and limit off-site impacts, though predicting rates based on harvest removals is complicated by atmospheric deposition, adsorption and losses through leaching, erosion, denitrification and volatilisation (Heilman and Norby, 1998).

Consider returning ash Return of ash could aid in replacing nutrients removed at harvest (other than N, which is volatilised during combustion). However, efficient means of distribution, and integration into the crop/forest fertiliser strategy, must be found to ensure net positive returns (in terms of both financial and GHG balance) from this practice.

Organic matter addition:

Apply additional organic matter in the form of manure, biosolids and recycled organic composts Recycled organics such as manures, biosolids and composts are more effective than fresh plant residues in raising soil C because the carbon is present as relatively more recalcitrant fractions (Inbar et al. 1990; Zinati et al., 2001;Poulton, 1995)

Species

Consider mixed species to maximise site productivity Each species has a different carbon allocation strategy that results in a different pattern, rate, quality and quantity of organic carbon input to the soil (Lugo and Brown, 1993). Mixed species planting can maximize biomass production, where species have facilitative rather than competitive interaction. For example, mixtures including nitrogen-fixing species (e.g. acacia with eucalypts (Bauhus et al., 2000), lupin with pine (Beets and Madgwick, 1988), and clover with pasture grasses (Ledgard, 1991)) commonly produce higher total biomass yields than monocultures of either species.

Nitrogen fixing vegetation, like fertilization, can enhance soil C due to the increase in fertility. In addition, N fixation seems to add more C to soils than fertilization, on average, perhaps due to the fact that organic matter (in the form of litter) is added along with the N (Johnson and Curtis, 2001).

Cultivation

Minimise cultivation disturbance to reduce mineralisation and erosion losses Minimising soil disturbance during site preparation may conserve soil carbon, particularly on erodible soils (Kort et al. 1998). For example, in a trial of site preparation methods for *Eucalyptus globulus* in Portugal, surface harrowing reduced soil C stock (to 40cm depth) by 25%, while deep ploughing reduced it by 40%, 18 months after treatment (Madeira et al., 1989). Site preparation for tree planting commonly involves ripping, often in conjunction with mounding, confined to the planting line. Ripping depth and size of mound can be minimised without jeopardising early growth in some soil types (Lacey et al., 2001) but mounding is clearly essential for successful plantation establishment in other soil types/landscape positions. Longer rotations, or coppicing, reduce the frequency of soil disturbance in forest systems.

Minimum tillage or no-till (NT) planting techniques increase soil carbon in cropping systems (eg Lal, 1997; Smith, et al. 2000b; Del Grosso et al., 2002; Tolbert et al., 2002;West and Marland, 2002), though in Australia positive impact of NT on soil carbon is restricted to wetter temperate regions (Dalal and Chan, 2001).

The most significant factor for promoting soil carbon is high plant growth rate. Therefore, management practices, including site preparation, should be designed to address site-specific growth limitations to the crop or plantation so as to ensure successful establishment and maximum growth rate.

6. Monitoring soil carbon changes

Soil C stock is notoriously difficult to quantify due to substantial spatial variability at fine and broad scale, however effective sampling, validation and accounting methods are available (McKenzie et al. 2000; Conant and Paustian, 2002; Palmer et al., 2002). Nevertheless, accurate measurement and monitoring of soil carbon, particularly to detect change in the short term (< 5-10 years), is prohibitively expensive for routine accounting of carbon sequestration in bioenergy projects, as hundreds of samples will be required to obtain acceptable accuracy (Garten and Wullschleger, 1999; Smith, 2004).

Modelling presents an alternative approach to estimating soil carbon change. As indicated above, soil carbon change is determined by the balance between plant inputs and soil C turnover rate, influenced by initial soil C status, soil type and climate; models developed from understanding of these interactions can predict soil C change. The RothC model of soil C dynamics is well proven in many environments (e.g. Coleman et al., 1997; Romanya et al., 2000). Linked with a model to estimate plant inputs, RothC can be utilised to estimate soil carbon change, as demonstrated below using the FullCAM model. In order to simulate soil C dynamics of a project, baseline soil C and environmental data are required. In addition to total soil organic carbon, a measure of the labile and recalcitrant fractions is necessary. A fractionation procedure developed by Skjemstad et al. (2004), based on sieving to determine particulate organic carbon and charcoal measurement by photo-oxidation and 13C NMR, allows quantification of soil C pools equivalent to the conceptual pools of the RothC model. With suitable calibration, mid-infra red analysis can provide a simple, cost-effective approximation of pool structure (Janik et al., 1998).

Therefore, it is recommended that accounting for soil carbon change in bioenergy projects be undertaken through a combination of measurement, to establish the baseline C stocks in each of the soil C pools, and modelling to estimate carbon dynamics over time. Models of plant growth and soil C dynamics have been calibrated for many crop/forest systems (e.g. Paul et al., 2003), though further work is needed to parameterise models for a broader range of environments and agricultural and forest systems, and to improve the ability of models to predict impacts of complex soil processes over the long term.

7. Modelling the Impact of soil carbon change on greenhouse gas balance of bioenergy systems

The FullCAM model of carbon dynamics (Richards, 2001) was employed to investigate the impact of soil C change on greenhouse gas balance of bioenergy systems. FullCAM links the process-based model of forest growth, 3PG (Landsberg and Waring 1997), with the forest carbon accounting model CAMFor (Brack and Richards, 2002), and the RothC model of soil organic matter turnover (Jenkinson et al., 1987, 1991). A parallel series of modules within FullCAM models agricultural systems, linking CAMAg (equivalent to CamFor) with RothC. The calculation of displacement of fossil fuel through substitution by bioenergy follows the approach of the GORCAM model (Schlamadinger and Marland, 1996). Input data required are monthly climatic data, site specific soil data, and management events.

FullCAM was used to simulate the long term GHG balance of bioenergy systems in comparison with conventional forestry systems, for three sites for which model parameterisation was available (Paul et al. 2003).

Three conventional forestry systems and associated bioenergy systems were modelled:

System 1 Short rotation Eucalypt plantation producing pulplogs

Reference: harvest residues decay on site, stems used for pulp

Bioenergy Case: harvest residues used for bioenergy, stems used for pulp

System 2 Radiata pine sawlog plantation

Reference: thinning and harvest residues decay on site, stems used for pulp and construction timber, excess mill residues burnt to waste

Bioenergy Case: thinning, harvest and excess mill residues used for bioenergy, stems used for pulp and construction timber

System 3 Eucalyptus sawlog plantation

Reference: thinning and harvest residues decay on site, stems used for construction timber, excess mill residues burnt to waste

Bioenergy Case: thinning, harvest and excess mill residues used for bioenergy, stems used for construction timber.

Table 1 lists site and management details for each system. Further detail of the sites, model parameterisation and model assumptions are given by Paul et al. (2002) and Paul et al. (2003). Harvest recovery, utilisation, service life and landfill decay data are derived from unpublished studies by D. Gardner and F. Ximenes. In each case where forest residues are recovered for bioenergy it is assumed that 70% of the branch biomass is removed, leaving 30% of branch biomass, 100% leaf mass, and 2-3% stem mass (representing the stump), as litter. Where sawn timber is produced, it is assumed that 3.5% of the carbon from mill residues is utilised to dry the timber. The bioenergy and corresponding reference systems each produce the same mass of pulp and/or timber products. The bioenergy system is based on theoretical calculations for co-firing biomass in a 500 MW black coal power plant (Cowie, unpublished). The assumed displacement factor is 0.83 tC avoided fossil emission per tC in biofuel. Emissions from establishment and harvest are assumed to be, respectively, 1.1 tCO2e ha^{-1} and 0.073 tCO2e per tC in biomass harvested. Processing and transport emissions are assumed to be 0.40 tCO2e per tC in biomass used for bioenergy.

[INSERT TABLE 1 HERE]

7.1 Calculation of net GHG balance

Net GHG balance for each system is determined from the carbon sequestration by the growing forest, plus the credit for avoided fossil fuel emissions, less the GHG emissions incurred in producing, processing and transporting the biomass, including indirect emissions from fertiliser manufacture. Non-CO_2 greenhouse emissions, specifically N_2O release after N fertiliser application and non-CO_2 from fossil fuel combustion, are included. The GHG balance of the bioenergy system is compared with the GHG balance of the traditional (reference) system to determine the net benefits of the bioenergy system.

Model results indicate that each of the bioenergy systems has lower soil C after 100 years than the corresponding reference system (Fig 1). These differences are largely due to relative declines in the resistant plant matter and humified pools. The difference is greatest in the short rotation eucalypt system, which shows a 35 t ha^{-1} increase in soil C under the reference system, and a 6 t ha^{-1} increase in the bioenergy case. The sawlog eucalypt reference system shows an initial decline followed by stabilisation of soil C. Under the bioenergy system, there is a continuing decline, and this system shows a loss of 35 tC ha^{-1} over 100 years. The soil C stock varies little over 100 years in the pine system, both in the reference and bioenergy cases.

[INSERT FIGURE 1 NEAR HERE]

Changes in the soil C pool are small compared with the accumulation of C in tree biomass over the first rotation, and the growing pools of products (Fig. 2). Over several rotations displaced fossil fuel carbon becomes the dominant pool, particularly in the sawlog eucalypt system, which has the highest proportion of removed biomass allocated to bioenergy rather than wood products due to low mill recovery for this species.

[INSERT FIGURE 2 NEAR HERE]

The net GHG balance of the bioenergy systems in comparison with the corresponding reference systems shows a significant benefit in increased C stocks for all three forestry systems (Fig. 3, Table 2). The displaced fossil fuel carbon is thirteen to twenty-two times greater than the relative decline in soil C stock.

[INSERT FIGURE 3 NEAR HERE]

8. Summary and Conclusion

Replacing current agricultural and forestry systems with systems that produce biomass for bioenergy is likely to affect soil carbon stocks, because it will alter the balance between organic matter inputs and losses from the soil carbon pool. There is a risk of depletion of soil carbon stocks in biomass production systems, because a higher proportion of the organic matter and nutrients are removed from the site, compared with conventional grain and timber production systems. Environmental and management factors will govern the magnitude and direction of change. Initial soil carbon content has a major influence: losses are most likely where soil C is initially high. Bioenergy systems such as coppiced willow, switchgrass, or long-rotation timber+biomass plantations, are likely to enhance soil carbon where these replace conventional cropping, as intensively cropped soils are generally depleted in soil C. Soil C losses are most likely where soil C is initially high, such as where improved pasture is converted to biomass production:

short –term loss of soil C is likely, and the equilibrium soil C stock under bioenergy systems may be lower than that of the previous pasture. Intensively managed bioenergy systems, such as perennial grasses and short rotation woody crops, are likely to have lower equilibrium soil C than long rotation forests, due to more frequent site disturbance and high rate of biomass removal. Measures that enhance soil C include maintenance of fertility through application of organic or artificial fertilisers or inclusion of legumes to promote plant growth, and retention of nutrient-rich foliage on-site.

Modelling results demonstrate that loss of soil C in bioenergy systems is associated with declines in the resistant plant matter and humified soil C pools. However, the relative soil C losses in bioenergy systems are likely to be small in comparison with displaced fossil fuel carbon, both for short rotation and sawlog plantations.

Thus, although there may be a small decline in soil C associated with biomass production, this is negligible in comparison with the contribution of bioenergy systems towards greenhouse mitigation through avoided fossil fuel emissions.

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Table 1 Site and management details for three conventional forestry systems and corresponding bioenergy systems.

System	Short rotation eucalypt		Pine		Sawlog eucalypt	
Species	*Eucalyptus globulus*		*Pinus radiata*		*Eucalyptus grandis*	
Location	Western Australia		South Australia		South-eastern Queensland	
Mean annual rainfall (mm)	1022		704		1138	
Mean annual air temperature (ºC)	14.9		13.4		20.4	
Soil type	Sandy loam		Sand		Sandy loam	
Initial soil C (tC ha^{-1} 0-30cm)	49.8		43.9		67.7	
Rotation length (years)	10		35		28	
Thinning (age, % biomass removed) Thinning 1 Thinning 2 Thinning 3	 NA NA NA		 10 (50) 24 (25) 27 (10)		 10 (50) 18 (50) NA	
Assumed fate of aboveground biomass (%)						
Activity	Reference	Bioenergy	Reference	Bioenergy	Reference	Bioenergy
Thinning 1 bioenergy litter	NA	NA	 0 100	 88 12	 0 100	 82 18
Thinning 2 pulp sawn timber bioenergy mill residue litter	NA	NA	 24 16 1 32 27	 24 16 48 1 11	 0 0 0 0 100	 0 0 0 82 18
Thinning 3 pulp construction timber bioenergy litter	NA	NA	 24 16 1 32 27	 24 16 48 1 11	NA	NA
ClearFall pulp sawn timber bioenergy mill residue litter	 50 0 0 0 50	 50 0 31 0 19	 3 29 1 45 22	 3 29 58 1 9	 0 10 0.4 22 68	 0 10 75 0.4 15

Table 2 Change in carbon stock of the forest, product and fossil fuel pools over 100 years.

Positive values indicate a gain, negative values a decline.

	Short rotation Eucalypt (tC ha^{-1})			Pine (tC ha^{-1})			Sawlog Eucalypt (tC ha^{-1})		
	Reference	Bioenergy	Difference[4]	Reference	Bioenergy	Difference[4]	Reference	Bioenergy	Difference[4]
Pool									
Soil[1]	35	6	-30	3	-5	-6	-9	-35	-19
Litter[2]	5	2	-2	2	1	0	5	2	-1
Trees[2]	76	76	0	62	62	0	96	96	0
Products in use[1]	18	20	0	38	39	0	26	30	0
Products in landfill[1]	228	228	0	41	41	0	21	21	0
Fossil fuel displaced by bioenergy[3]	0	391	391	0	165	165	0	594	594
Fossil fuel spent	-16	-78	-62	-7	-29	-22	-17	-96	-79
Net GHG balance	**345**	**644**	**296**	**139**	**274**	**136**	**123**	**612**	**494**

[1] Value at 100 years determined from fitted trend line, to overcome influence of fluctuating pool size.

[2] Value at 100 years determined from average carbon stock of pool over the period.

[3] Total carbon stock of pool at 100 years.

[4] Difference between bioenergy and reference case at 100 years.

Figure 1 Soil carbon pools for short rotation eucalypt (1.1), pine (1.2) and sawlog eucalypt (1.3) conventional forestry systems (a) and bioenergy systems (b).

Figure 2 Carbon stock of all forest and product pools, avoided emissions, and fossil fuel spent for each bioenergy system for (a)short rotation eucalypt, (b) pine and (c) sawlog eucalypt. Fossil fuel spent is a negative value. The net carbon stock is indicated by the broken line.

Figure 3 Difference between the bioenergy and reference cases in carbon stock of each pool in (a) short rotation eucalypt, (b) pine and (c) sawlog eucalypt. For soil and litter pools negative values indicate a relative decline in carbon stock; for fossil fuel spent, negative values indicate greater emissions.