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Implications of land-use change to Short Rotation Forestry in Great Britain for soil and biomass carbon

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Abstract

Land-use change can have significant impacts on soil and aboveground carbon (C) stocks and there is a clear need to identify sustainable land uses which maximize C mitigation potential. Land-use transitions from agricultural to bioenergy crops are increasingly common in Europe with one option being Short Rotation Forestry (SRF). Research on the impact on C stocks of the establishment of SRF is limited, but given the potential for this bioenergy crop in temperate climates, there is an evident knowledge gap. Here, we examine changes in soil C stock following the establishment of SRF using combined short (30 cm depth) and deep (1 m depth) soil cores at 11 sites representing 29 transitions from agriculture to SRF. We compare the effects of tree species including 9 coniferous, 16 broadleaved and 4 *Eucalyptus* transitions. SRF aboveground and root biomass were also estimated in 15 of the transitions using tree mensuration data allowing assessments of changes in total ecosystem C stock. Planting coniferous SRF, compared to broadleaved and *Eucalyptus* SRF, resulted in greater accumulation of litter and overall increased soil C stock relative to agricultural controls. Though broadleaved SRF had no overall effect on soil C stock, it showed the most variable response suggesting species-specific effects and interactions with soil types. While *Eucalyptus* transitions induced a reduction in soil C stocks, this was not significant unless considered on a soil mass basis. Given the relatively young age and limited number of *Eucalyptus* plantations, it is not possible to say whether this reduction will persist in older stands. Combining estimates of C stocks from different ecosystem components (e.g., soil, aboveground biomass) reinforced the accumulation of C under coniferous SRF, and indicates generally positive effects of SRF on whole-ecosystem C. These results fill an important knowledge gap and provide data for modelling of future scenarios of LUC.

Keywords: afforestation, bioenergy, coniferous, deciduous, eucalypt, land use, organic carbon, SRF

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Introduction

Land use and land management are dominant factors which influence soil carbon (C) and Greenhouse Gas (GHG) dynamics over the long term. Conversion of natural habitats to agricultural land use may severely deplete soil C within only a few years (Davidson & Ackerman, 1993; Murty *et al.*, 2002; Don *et al.*, 2012), while it can take decades to recover former levels of soil C following reversion of agricultural to extensive land use (Paul *et al.*, 2002; Laganière *et al.*, 2010; Poeplau *et al.*, 2011). Consequently, there is a strong interest to identify more sustainable land uses which can increase C sequestration and maximize mitigation potential (Lal, 2004). The transition from 'traditional' agricultural use to bioenergy crops is an increasingly common land-use change in Europe, therefore, understanding the associated impact on the soil and site C balance is essential (Don *et al.*, 2012).

Bioenergy land use has the potential to offset anthropogenic GHG atmospheric increases though C sequestration via increased soil organic carbon (SOC) stocks, and bring wider environmental benefits, but there still remain concerns about its sustainability (Cowie *et al.*, 2006; Field *et al.*, 2008; Rowe *et al.*, 2009). While impacts of bioenergy production on food production and food security are complex and still unresolved, it is evident that in some contexts planting bioenergy crops may result in changes in soil C stocks (Cowie *et al.*, 2006; Smith *et al.*, 2013). Identifying scenarios (e.g., particular combinations of climate, soil type and crop) where changes in soil C are positive rather than negative is of great importance in

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developing land-management strategies which maximize the C mitigation potential. This is particularly a key for bioenergy as modelled scenarios, based on EU policy options, suggest that conversion into bioenergy crops provides the greatest potential for C mitigation (Smith *et al.*, 2000). In addition, biomass energy sources are seen as an important resource if targets for renewable energy production in the United Kingdom are to be met, thus there is growing urgency to ensure that policies provide maximum C mitigation (Grogan & Matthews, 2001; Ostle *et al.*, 2009; Rowe *et al.*, 2009; Aylott *et al.*, 2010; Smith *et al.*, 2013).

Short rotation forest (SRF) is one potential bioenergy crop which is suitable for temperate climates (McKay, 2011; Leslie *et al.*, 2012). In contrast to traditional forestry, SRF plantations have a higher density (>2500 tree ha^{-1}) and shorter rotations (generally 8–20 years) (McKay, 2011) with harvesting occurring when diameter at breast height (DBH) reaches around 20 cm. This silvicultural system is particularly suitable for bioenergy crop applications as it provides relatively high yields over short time frames (Proe *et al.*, 2002; Hoffmann & Weih, 2005; Hardcastle *et al.*, 2006; McKay, 2011). SRF could provide added flexibility to the woody bioenergy supply in the United Kingdom as, unlike coppice crops, harvesting can take place year round, and the product has a lower bark and moisture content and a higher density making it an ideal fuel source (Hardcastle *et al.*, 2006; Leslie *et al.*, 2012). Biomass yields of SRF may also be higher than coppice systems per unit area (McKay, 2011).

Despite the interest in SRF systems, there is limited data on changes in soil C caused by their establishment in temperate forestry, particularly in Europe (Vanguelova & Pitman, 2011). Concern has also been raised that SRF may lead to long-term nutrient depletion and acidification in soils (Vanguelova & Pitman, 2011). Clearly, providing an evidence base of the impacts of SRF on soils is a key component in understanding the long-term sustainability of SRF and, if appropriate, in developing support for this system from policy makers, potential end-users and the public (Hardcastle *et al.*, 2006; McKay, 2011).

Vanguelova & Pitman (2011) identified a need to compare the effects of different tree species and rotation lengths on C sequestration under SRF land use. Coniferous and broadleaved species can both be grown in SRF systems, with current UK SRF research focused on species with high-growth rates (McKay, 2011). In addition to changes in soil C, planting of SRF will also influence total ecosystem C stock through changes in standing aboveground biomass, root biomass and litter (Hoffmann & Weih, 2005; Hardcastle *et al.*, 2006; McKay, 2011). These C pools fluctuate with season and stage in the harvest cycle, but changes in aboveground and root biomass have been found to be larger than changes in soil C, especially in young plantations (Laclau, 2003; Guo *et al.*, 2008). Time since establishment is therefore expected to have a strong influence on the status of different C stocks.

In this study, utilizing a paired-site approach, we assess the impacts on litter and soil C of 11 different tree species currently being grown in SRF trials or commercial systems with analogous characteristics. Those studied include coniferous, broadleaved and eucalypt species as the three main tree types under consideration (Hoffmann & Weih, 2005; Hardcastle *et al.*, 2006; McKay, 2011; Leslie *et al.*, 2012). The paired-site approach makes a comparison between a target land use and an adjacent original land use; it has been used widely to examine changes in soil C (Davis & Condron, 2002; Laganière *et al.*, 2010). It is a valuable approach, particularly to examine longer term changes, since a retrospective or repeated measure sampling design is often not possible. However, it is necessary to ensure that the comparison of land uses is valid with no pre-existing differences in land use, soil type, topography etc. (Wellock *et al.*, 2011; Hewitt *et al.*, 2012). Changes in bulk density across paired-sites or transitions may result in bias in C stock estimates with fixed-depth sampling due to differences in sampled soil mass (Gifford & Roderick, 2003; Wellock *et al.*, 2011). Consequently, we also use a mass-based approach to ensure that any differences in soil C are not largely an artefact of differences in bulk density. Furthermore, measurements of soil C are extended to 1 m depth, thus providing opportunity to consider SRF effects deeper in the soil profile and in 18 of the 29 transitions studied, above- and belowground biomass C stocks were estimated. We combine these assessments to provide the first comprehensive assessment of the impacts on ecosystem C stocks associated with SRF production in GB.

Materials and methods

Site selection

We identified a range of potential paired-sites for sampling that were either located within replicated experimental plots or under commercial forest stands. Subsequently, sampling was conducted at 11 sites across the United Kingdom, with each having suitable SRF species and an adjacent paired control (Table 1 and Table S1). Existing data on land management history and soil type (Table 1) for these sites were used to ensure that comparisons between control and SRF land uses were appropriate. In addition, photographs of deep cores (see below; Fig. S1) and soil texture data from short cores (Table 1) were also examined following sampling to check that soils were

Table 1 Details and soil characteristics of sampling locations used to examine the effects of Short Rotation Forestry on carbon stocks in GB. Transition units in bold represent control land use. Soil type based on the Avery soil classification; texture class derived based on the Soil Survey of England & Wales texture classes. C stock represent means ± SD; n = 15

						Texture (0–30 cm)				C stock (0 – 30 cm)
Site no.	Region	Lat.	Long.	Transition unit	Soil type	% Clay	% Silt	% Sand	Texture class	tC ha^{-1}
1	North-East, England	55.2	−1.5	**Arable**	Brown earth	4.7	35.6	59.7	Sandy loam	111.9 ± 16.3
				E. gunnii	Brown earth	5.0	38.9	54.6	Sandy loam	98.6 ± 14.0
				E. nitens	Brown earth	4.7	35.1	60.2	Sandy loam	93.0 ± 14.7
2	Powys, Wales	52.0	−3.6	**Pasture**	Brown earth	9.2	75.3	15.5	Silt loam	76.2 ± 9.0
				H. Larch	Brown earth	10.2	75.6	14.2	Silt loam	76.3 ± 8.4
				Sycamore	Brown earth	7.7	73.7	18.6	Silt loam	65.1 ± 7.3
3	North-East, England	54.3	−0.5	**Rough grass**	Surface-water gley	3.7	43.6	52.7	Sandy loam	101.8 ± 29.3
				Alder	Surface-water gley	4.3	42.5	53.2	Sandy loam	91.4 ± 39.4
				Scots pine	Surface-water gley	3.9	36.6	59.5	Sandy loam	90.9 ± 33.8
				Silver birch	Surface-water gley	4.2	43.8	52.0	Sandy loam	99.1 ± 22.1
				Beech	Surface-water gley	4.3	47.6	48.1	Sandy silt loam	97.5 ± 33.5
4	East Midlands, England	53.34	−1.0	**Rough grass**	Brown earth	2.9	28.0	69.1	Sandy loam	54.0 ± 13.6
				E. gunnii	Brown earth	2.0	21.0	77.0	Loamy sand	40.8 ± 14.2
				E. nitens	Brown earth	2.4	25.2	72.4	Sandy loam	45.5 ± 13.8
5	Moray, Scotland	57.6	−3.2	**Rough grass**	Podzol	4.1	39.8	57.5	Sandy loam	94.8 ± 22.4
				Downy birch	Podzol	2.2	40.1	57.7	Sandy loam	111.5 ± 31.4
				Silver birch	Podzol	1.7	34.1	64.2	Sandy loam	81.5 ± 21.3
				Sitka spruce	Podzol	2.7	42.5	54.8	Sandy loam	136.9 ± 44.5
6	Moray, Scotland	57.7	−3.3	**Pasture**	Ground-water gley	0.7	18.4	80.9	Loamy sand	39.3 ± 8.5
				Poplar	Ground-water gley	0.1	15.8	84.1	Loamy sand	35.2 ± 6.2
				Alder	Ground-water gley	0.4	14.7	84.9	Loamy sand	38.8 ± 8.5
				Ash	Ground-water gley	0.1	15.7	84.2	Loamy sand	35.6 ± 6.6
7	North-West, England	54.0	−2.4	**Rough grass**	Surface-water gley	4.8	58.1	37.1	Sandy silt loam	117.2 ± 46.3
				Alder	Surface-water gley	5.9	50.9	43.2	Sandy silt loam	122.3 ± 25.7
				Scots pine	Surface-water gley	5.3	48.9	45.8	Sandy silt loam	146.8 ± 45.7
				Sitka spruce	Surface-water gley	5.3	51.3	43.4	Sandy silt loam	143.4 ± 43.7
8	Aberdeenshire, Scotland	56.9	−2.6	**Pasture**	Podzol	3.2	47.1	49.7	Sandy silt loam	80.6 ± 9.9
				Sycamore	Podzol	4.2	48.4	47.4	Sandy silt loam	83.1 ± 14.5
				Scots pine	Podzol	4.3	50.3	45.4	Sandy silt loam	76.2 ± 20.9
				H. Larch	Podzol	5.3	50.6	44.1	Sandy silt loam	74.5 ± 13.1

(continued)

Table 1 (continued)

Site no.	Region	Lat.	Long.	Transition unit	Soil type	Texture (0–30 cm) % Clay	% Silt	% Sand	Texture class	C stock (0 – 30 cm) tC ha^{-1}
9	North Lanarkshire, Scotland	55.8	−3.8	**Pasture**	Surface-water gley	4.3	53.3	42.4	Sandy silt loam	122.9 ± 24.1
				Alder	Surface-water gley	6.1	64.7	29.2	Sandy silt loam	100.8 ± 25.0
				Poplar	Surface-water gley	3.6	67.6	28.8	Sandy silt loam	92.0 ± 10.7
				Sitka spruce	Surface-water gley	4.6	59.4	36.0	Sandy silt loam	140.9 ± 27.8
10	North-West, England	54.7	−2.8	**Pasture**	Surface-water gley	4.1	51.6	44.3	Sandy silt loam	83.0 ± 12.2
				Ash	Surface-water gley	3.5	48.9	47.6	Sandy silt loam	89.5 ± 11.9
				Sycamore	Surface-water gley	3.5	52.9	43.6	Sandy silt loam	85.8 ± 9.2
				Alder	Surface-water gley	3.1	53.9	43.0	Sandy silt loam	92.4 ± 13.5
11	Fife,	56.1	−3.6	**Rough grass**	Surface-water gley	4.8	57.9	37.3	Sandy silt loam	83.2 ± 9.6
	Scotland			Scots pine	Surface-water gley	4.9	53.0	42.1	Sandy silt loam	72.5 ± 16.0

comparable. Focal tree species were selected based on current literature and expert advice regarding the most promising species for commercial use (Proe *et al.*, 2002; Hardcastle *et al.*, 2006; McKay, 2011). These included Alder (*Alnus incana and A glutinosa*), Ash (*Fraxinus excelsior*), Cider gum (*Eucalyptus gunni*), Downy birch (*Betula pubescens*), Hybrid larch (*Larix x eurolepis*), Poplar (*Populus spp.*), Scots pine (*Pinus sylvestris*), Shining gum (*Eucalyptus nitens*), Silver birch (*Betula pendula*), Sitka spruce (*Picea sitchensis*) and Sycamore (*Acer pseudoplatanus*), which were broadly classified into coniferous, broadleaved and eucalypt for the purpose of analysis and inference.

Site 3 was established in the 1950s and had not been harvested but the DBH of the broadleaved tree species within this site was still within the range expected for SRF. Consequently, this site was considered to provide useful comparisons for younger and second rotation transitions under different SRF types. Site 7 was also established in the 1950s, but is now in a second rotation of SRF after being harvested and replanted in 1991, while all other sites are still in first rotation (Table S1). The mean age of the remaining transitions was 14.6 years (ranging from 4 to 24 years).

Approach and sampling method

To allow robust sampling and statistical analysis, replicated experimental plots under the same land use (i.e. tree species or control) were treated as one continuous area. For clarity, these continuous areas of the same species are hereafter referred to as 'transition units'. These 'transition units' were sampled using a hierarchical design, developed to capture variability across different spatial scales (Conant & Paustian, 2002; Conant *et al.*, 2003). This consisted of five sampling plots randomly located across each transition unit within which individual soil cores were taken from three neighbouring positions (within 1–2 m). This gave a total of 15 spatially nested samples per transition unit, accounting for both field-scale (sampling plots) and plot-scale (core within plots) variability. The five sampling locations per transition unit were randomly selected from intersections of a grid overlaid on a map of the transition unit. The resolution of the grid was adjusted to ensure that there were a minimum of 50 grid intersections, with the condition that the resolution of the grid could not be less than 5 m. A 20 m perimeter buffer was also used to reduce potential edge effects. Although in some cases, smaller experimental plots required that this outer buffer to be reduced (to a minimum of 5 m) to attain the 50 grid intersects necessary.

Short cores (0–30 cm). The three within-plot soil samples were taken using a split tube soil sampler with an inner diameter of 4.8 cm (Eijkelkamp Agrisearch Equipment BV, Giesbeek, the Netherlands) to a depth of 30 cm. The first core was taken at the grid intersect, with two further cores taken at distances of 1 m and 1.5 m in random compass directions from the intersect. Before each core was taken, litter (L) and fermentation (L_f) horizons were also collected from a 25 cm × 25 cm area centred on the coring location. Soil cores were divided in the field into 0–15 cm and 15–30 cm (measuring from the base of the core), individually bagged and returned to the laboratory.

Deep cores (0-–100 cm). One of the five sampling plots was randomly selected and three metre cores were taken following the same spacing as the 30 cm core, with the exact coring locations adjusted to avoid those of the 30 cm cores. Cores were taken using a window sampler system with a 4.4 cm cutting diameter to a depth of 1.3 m (Eijkelkamp Agrisearch Equipment BV), allowing full 1 m core to be taken in one section (Fig. S1). The corer was hammered into the soil using a Cobra pneumatic petrol breaker (Atlas Copco, UK) and then extracted from the soil using a proprietary lever system (Eijkelkamp Agrisearch Equipment BV). Once extracted, the soil core was removed intact in a protective plastic sleeve, labelled and placed in PVC piping for transport back to the laboratory. If coring to the full depth was not possible, for example when large stones or bedrock were encountered, a tape measure was used to determine the precise depth of the cored hole, and the reduced length of the core was recorded.

Laboratory processing

Litter samples were dried at 80 °C for 24 h and dry mass of woody material (e.g., twigs, branches), and leaves/needles and undifferentiated material, were recorded. Litter was assumed to have C concentration of 45% for estimation of litter C stocks.

Short cores (0–30 cm). The fresh mass of the 0–15 cm and 15–30 cm core sections was recorded and sections were then cut lengthways into quarters, for separate subsequent analyses. One quarter, together with all the stones and roots (>2 mm), was separated from the remaining sections and processed for soil C and bulk density (BD). The fresh mass was recorded and then the samples were air-dried at 25 °C for minimum of 10 days. Air-dried samples were reweighed, sieved to 2 mm and the mass and volume of stones and roots remaining on the sieve recorded. A subsample of the sieved soil (15–18 g) was oven-dried (105 °C for 12 h) and moisture loss recorded. The oven-dried subsample of soil was ball-milled (Fritsch Planetary Mill) to a fine powder, and then a 100 mg subsample was used for the assessment of C and N concentration using an elemental analyser (Leco, truspec CN).

Bulk density (Table S2) was calculated using values of moisture loss following methods in the GB Countryside Survey (Emmett *et al*., 2008; Reynolds *et al*., 2013). These calculations omitted the mass and volume in the soil cores accounted for by stones, and so are corrected to represent the fine earth proportion (Schrumpf *et al*., 2011). The UK Countryside Survey (CEH) conducted a pilot study to compare different protocols to estimate BD in different soil types and found that the method used in this study was consistent with other protocols and within the ranges of typical values expected for each of the soil types (Emmett *et al*., 2008).

The soil C concentration and bulk density data were used to derive mass-based values of soil C stock to account for differences in bulk density across transitions. A soil C stock was calculated by interpolation of cumulative soil mass (CSM) values, using a reference dry soil mass of 300 kg m^{-2}, following the method of Gifford & Roderick (2003).

A second quarter was used for measurement of soil pH, with soil core sections from each sampling plot bulked to give 10 composite samples per transition unit (five each for the 0–15 cm and 15–30 cm depths). The fresh, bulked samples were sieved to 4 mm to remove stones and roots. Ten gram of bulk soil was then mixed well with 25 mL of deionized water and allowed to stand for 30 min, before the pH of the liquid layer was recorded (Hanna pH210 Meter). Proportions of sand, silt and clay were analysed by laser diffraction (Malvern Mastersizer 2000) in 0–15 cm and 15–30 cm depths from samples bulked across plots.

Deep cores (0-100 cm). On return to the laboratory, the metre cores were divided into four sections: 0–15, 15–30, 30–50 and 50–100 cm. Some compression of the soil cores was often observed, resulting in a recovered core length of less than 100 cm. In most cases, the compression was slight (<10 cm) and appeared to be restricted mainly to the upper 15 cm of the core. In these cases, compression was assumed to be restricted to the upper 15 cm and subdivision of the core measured from the bottom. In a few cases, compression of greater than 10 cm was apparent. In all these cases, it was evident that compression was taking place throughout the length of the core. This was demonstrated by bulging that was clearly evident in places along the entire core length when removing the plastic sleeve. In these cases, the compression was therefore assumed to be evenly distributed through the core, and proportional reductions in all the core sections were made. Each core section was divided lengthways and half the core and all root and stones processed for bulk density and carbon content as outlined for the 30 cm cores.

Mensuration data

Aboveground and root biomass C estimates were derived from mensuration data at six sites (representing 15 transitions), so that a whole-ecosystem assessment of C stocks across SRF bioenergy transitions could be made. Standard forest mensuration techniques were used to measure tree height and diameter at breast height which were then entered into known relationships relating these to total biomass and for estimating C stock of different tree species (see Supporting information). For purposes of comparison, arable and grassland controls were assigned an aboveground biomass value of 1 tC ha^{-1} following Milne & Brown (1997).

Statistical methods

The influence of SRF transitions on litter and soil C variables, and soil pH, was tested using linear mixed-effect models with the *lme* package in the R statistical program (R Development Core Team, 2011). The significance of these models was examined using a likelihood ratio test between a null model including only random terms and the chosen models. Firstly, the overall effect of planting SRF on these variables was tested, with a dummy variable contrasting control and SRF transition units entered as a fixed effect. Secondly, the effect of the different land uses (control and SRF types) was tested, with a fixed effect containing levels for Control, coniferous, broadleaved and eucalypt transition units. Means and standard errors reported are derived from linear mixed models with site, and field nested within site, and plot nested within field, entered as random effects in all models to ensure that appropriate comparisons of transition units were accounted for within site. The significance of differences between agricultural controls and SRF, or SRF types, was tested using Tukeys multiple comparison in the *glht* function in the *multcomp* package (Hothorn *et al.*, 2008). Simple R^2 values for these fixed effects were derived as the percentage of residuals accounted for by the fixed effect compared to a null model that includes only the random nested terms (Xu, 2003). Data on litter mass and soil C were log-transformed prior to testing where necessary to meet model assumptions. Site 3 was not included in these analyses due to the extended time since establishment without harvesting.

The relationship between time since SRF establishment and changes in soil C, and changes in the standing litter layer and aboveground biomass, was examined using linear models to assess temporal patterns. In addition, differences in the rates of change in soil C and aboveground biomass, and total ecosystem C stock, between tree types were tested using a nonparametric Kruskal–Wallis test.

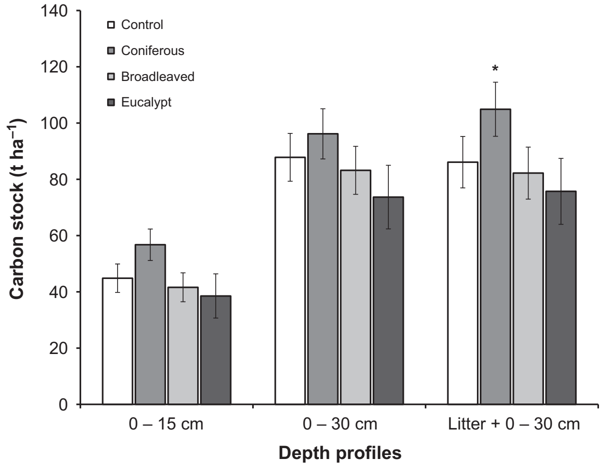

Fig. 1 Estimated carbon stocks (tC ha^{-1}) in different litter/soil depth profiles (to 30 cm depth) under transition land uses (controls and different types of short-rotation forestry). Means and standard errors derived from linear mixed models with site and field within site as random effects. Asterisk indicates tree type is significantly different from the control.

Results

Litter and soil C

Litter mass was higher under SRF (8.02 ± 1.40 tC ha^{-1}) compared to the control (0.97 ± 1.93 tC ha^{-1}) [$\chi^2(1)$ = 24.9, $P < 0.001$, R^2 = 36.3%]. Litter mass also varied between the land uses [$\chi^2(3)$ = 39.6, $P < 0.001$, R^2 = 48.9%] with posthoc testing confirming that coniferous (14.31 ± 1.73 tC ha^{-1}, $P < 0.001$), eucalypt (8.60 ± 2.50 tC ha^{-1}, $P < 0.001$) and broadleaved (3.83 ± 1.43 tC ha^{-1}, $P < 0.001$) tree types had higher litter mass than the agricultural control. Litter mass was also significantly greater under coniferous than broadleaved land use ($P < 0.001$).

There was no overall difference in the soil C stock between control (44.86 ± 5.28 tC ha^{-1}) and SRF (45.80 ± 4.67 tC ha^{-1}) at the 0–15 cm depth (in the short cores)[$\chi^2(1)$ = 0.06, P = 0.805, R^2 = 0.08%]. This was also the case between control (87.81 ± 8.75 tCha^{-1}) and SRF (85.10 ± 8.38 tC ha^{-1}) with summed 0–30 cm depth [$\chi^2(1)$ = 0.47, P = 0.490, R^2 = 0.045%]. Furthermore, including litter layer C did not differentiate control and SRF [$\chi^2(1)$ = 0.06, P = 0.809, R^2 = 0.1%].

Differences between land uses (e.g., controls and different SRF types), however, were apparent at the 0–15 cm depth [$\chi^2(3)$ = 10.16, P = 0.017, R^2 = 9.41%]. Posthoc testing showed that, though there were no significant differences between the control and any SRF type, coniferous and broadleaved SRF significantly differed (P = 0.013; Fig. 1). Likewise, while there was a significant effect of land use on soil C stock in the summed 0–30 cm [$\chi^2(3)$ = 8.56, P = 0.036, R^2 = 5.90%;], there were no significant differences between control and any SRF, but coniferous differed from eucalypt SRF (P = 0.035; Fig. 1). Including litter layer C resulted in a stronger effect of land use [$\chi^2(3)$ = 15.51, P = 0.001, R^2 = 10.58%; Fig. 1], with significant differences between control and coniferous (P = 0.048; Fig. 1), and also between coniferous and broadleaved (P = 0.004; Fig. 1) and eucalypt SRF (P = 0.022; Fig. 1).

Land use also had a significant effect on soil C stock using the CSM mass-based approach [$\chi^2(3)$ = 10.92, P = 0.012, R^2 = 8.57%]. Though not directly comparable to the fixed depth means, the relative levels of soil C stock between land uses were maintained with coniferous (106.28 ± 13.60 tC ha^{-1}) > control (95.39 ± 13.17 tC ha^{-1}) > broadleaved (90.27 ± 13.27 tC ha^{-1}) > eucalypt (81.52 ± 15.47 tC ha^{-1}). Posthoc tests showed a significant difference between control and *Eucalyptus* (P = 0.016), and also between *Eucalyptus* and coniferous (P = 0.020).

Soil C densities in the deep cores (to 1 m) supported the patterns between SRF types demonstrated in the short cores. At 0–30 cm, there were significant differences between control and coniferous SRF (P = 0.019; Table 2), and also between coniferous and broadleaved (P = 0.002; data not shown), and coniferous and *Eucalyptus* (P = 0.002; data not shown). Significant differences in soil C stock between control and coniferous was maintained as the profile depth increased (Table 2). However, the differences between control and coniferous in individual sections were significant at 30–50 cm (P = 0.001), but not at 50–100 cm (P = 0.989).

Table 2 Change in soil carbon stock (tC ha^{-1}) under different types of short rotation forestry from 1 m cores. Values represent estimated differences from Tukey contrast of treatments in mixed model; positive values indicate SRF has greater C stocks than controls; $P < 0.001$ = ***, $P < 0.01$ = **, $P < 0.05$ = *, ns = not significant

	Soil depth		
Land use contrast	0–30 cm	0–50 cm	0–100 cm
Control to Coniferous	+16.12*	+35.97 ***	+43.39**
Control to Deciduous	-2.52^{ns}	-7.29^{ns}	-14.06^{ns}
Control to Eucalypt	-18.67^{ns}	-26.26^{ns}	-36.81^{ns}

Ecosystem C stocks and rates of change in soil and aboveground biomass C

There was no clear relationship between time since SRF establishment and estimated aboveground biomass C ($P = 0.287$; Fig. 2a). However, it is evident that stocks of aboveground biomass C are, in some cases, as large as stocks of soil C to 30 cm depth (Fig. 3) and that coniferous trees species have a higher level of estimated aboveground biomass C than broadleaved tree species (Fig. 2a, Fig. 3). Including these estimates highlights that aboveground biomass C is a significant component of SRF C stocks, and further enhances the positive effect of coniferous SRF on C stocks (Fig. 3). Percent change in soil C stock also showed no linear relationship with time since establishment ($P = 0.780$; Fig. 2b). Site 3, which had not undergone harvest, had comparable aboveground C stocks to Scots pine in second rotation at Site 7, but not changes in soil C stock (Fig. 2).

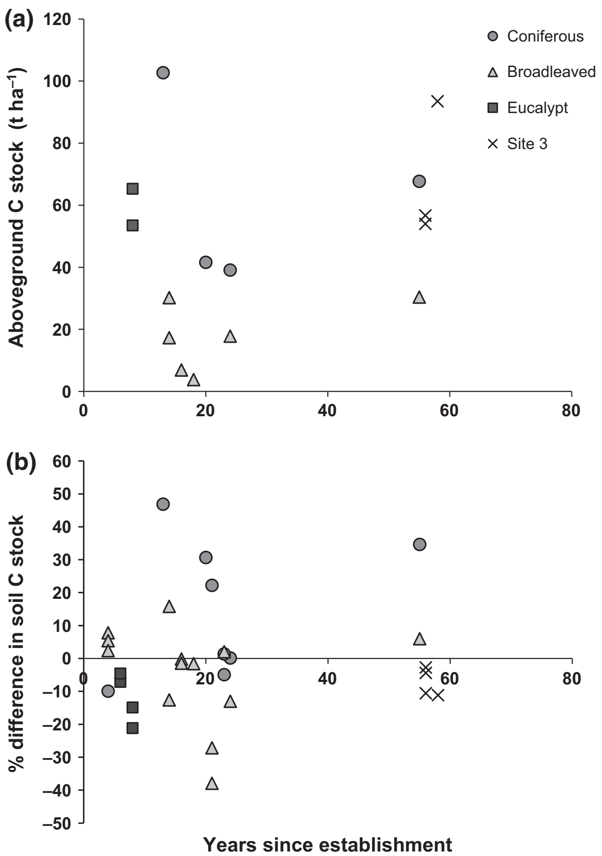

Fig. 2 Relationship between years since SRF establishment and (a) above ground biomass tree C and (b) percent change in soil C stock (to 30 cm depth), across sites.

The estimated annual rate of change in soil C at 0–30 cm was significantly different between tree types [Kruskal–Wallis $\chi^2(2) = 10.85$, $P = 0.004$]. Average changes under coniferous SRF was 0.29 ± 0.54 tC ha^{-1} yr^{-1} compared to 0.07 ± 0.24 tC ha^{-1} yr^{-1}under broadleaved SRF and −1.90 ± 0.22 tC ha^{-1} yr^{-1} decrease under eucalypt SRF (Table 3). Omitting the youngest transitions (4 years old) resulted in average changes of 0.70 ± 0.41 tC ha^{-1} yr^{-1}under coniferous and −0.28 ± 0.18 tC ha^{-1} yr^{-1} under broadleaved SRF, with tree type still significantly different [Kruskal–Wallis $\chi^2(2) = 8.41$, $P = 0.015$]. Likewise, the estimated rate of change in aboveground biomass C was significantly different between tree types [Kruskal–Wallis $\chi^2(2) = 8.68$, $P = 0.013$] and was greatest for eucalypts (Table 3).

Soil pH

Soil pH was lower under SRF than control land use in most sites, but this was not the case for all tree species (Table S2). Soil pH in the 0–15 cm depth was lower under SRF (5.67 ± 0.23) compared to the control [6.08 ± 0.24; $\chi2(1) = 9.04$, $P = 0.003$, $R^2 = 8.36\%$]. Although slightly reduced, this difference was also maintained at 15–30 cm depth with soil pH lower under SRF (5.96 ± 0.20) than the agricultural control (6.22 ± 0.22) [$\chi2(1) = 5.16$, $P = 0.023$, $R^2 = 4.61\%$].

The test of land uses at 0–15 cm depth indicated significant differences in soil pH [$\chi2(3) = 10.46$, $P = 0.015$, $R^2 = 17.49\%$] with posthoc testing showing differences between the agricultural control (6.09 ± 0.23) and coniferous (5.51 ± 0.25; $P = 0.012$). Under the remaining SRF types, pH was generally lower than the control but the differences were not significant under either eucalypt (5.58 ± 0.32, $P < 0.001$) or broadleaved (5.57 ± 0.28, $P < 0.001$) tree types. At 15–30 cm, despite overall differences between control and SRF, there were no significant differences in soil pH between the individual land uses [$\chi2(3) = 5.46$, $P = 0.141$, $R^2 = 1.25\%$]. Though mean values under the SRF species were generally lower than the controls (Control: 6.22 ± 0.22, coniferous: 5.96 ± 0.24, broadleaved: 6.00 ± 0.23, eucalypt: 5.85 ± 0.31), error terms showed noticeable overlap.

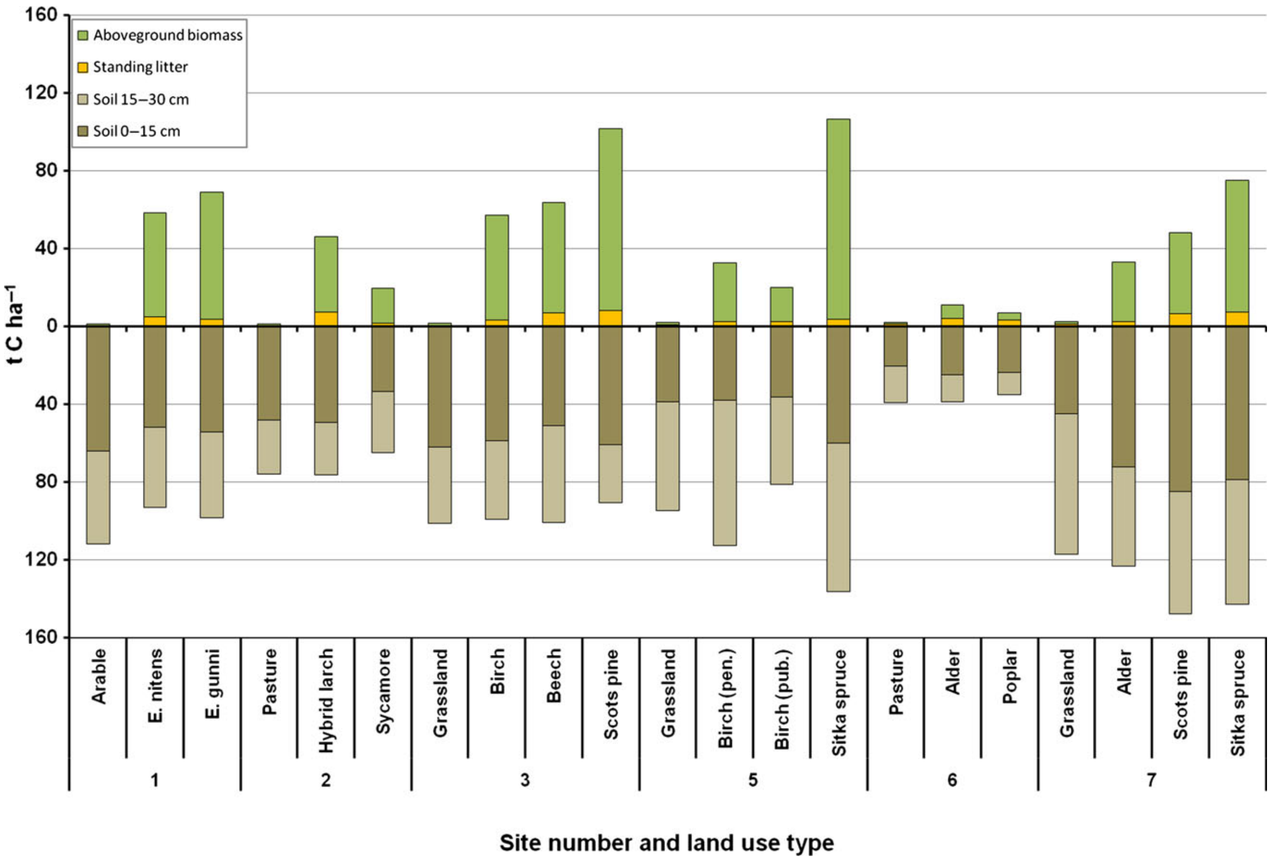

Fig. 3 Above and below ground carbon stocks (to 30 cm depth) at selected SRF sites. Site numbers follow that in Table 1.

Discussion

Given the potential for SRF in temperate climates and limited data, there is clearly a need to better understand the impacts associated with planting SRF on the C balance under such land-use change scenarios. This study provides the first detailed dataset of C stock changes across relevant SRF land-use transitions in GB.

Short Rotation Forestry, taken as a whole, had little overall effect on soil C. The range of transition ages may be expected to have increased variability across site C stocks, thereby masking any detectable difference. For example, higher variability in soil C content in young plantations is apparent across chronosequences under traditional forestry practices (Thuille & Schulze, 2006; Vesterdal *et al.*, 2007). In studies of conventional forestry practices, establishment of plantations on agricultural land has been found to result in an initial loss of soil C, followed by a slow recovery, with a net deficit lasting 10 or more years. (Paul *et al.*, 2002; Thuille & Schulze, 2006; Guo *et al.*, 2007). Though such a deficit was apparent in some young transitions, it was not consistent and there was no clear relationship between time since SRF establishment and soil C stock. Other studies using a paired-site approach across a range of tree species and soil types have also found both increases and decreases in soil C stock following afforestation (Wellock *et al.*, 2011; Hewitt *et al.*, 2012). In New Zealand, Hewitt *et al.* (2012) showed that coniferous forest soil, mainly under *Pinus radiata*, contained significantly less C than in low productivity grasslands, but several sites remained unchanged or even increased. Wellock *et al.* (2011) measured soil C density in paired-sites across Ireland under various coniferous and broadleaved tree species, but there was no significant overall change, despite a reduction in soil C density under forest. It is likely that increases and decreases across particular SRF transitions also cancel one another as stronger patterns emerged when considering different types of SRF tree species (i.e. coniferous, broadleaved and eucalypts).

SRF types

Lower stocks of soil C were recorded under the eucalypt SRF compared to the control, as demonstrated by both fixed depth and CSM assessment and by the negative estimated annual change of 1.90 tC ha^{-1} yr^{-1} in the 0–30 cm depth. Soil C stock under eucalypt SRF was

Table 3 Estimated rates of change in soil carbon (0–30 cm) and aboveground biomass carbon stocks in Short rotation forestry transitions in Great Britain. NA = data not available

Site	Species	Years since transition	Aboveground C change tC ha^{-1} yr^{-1}	Soil C change tC ha^{-1} yr^{-1}
	Broadleaved			
3	Alder	56	NA	−0.19
6	Alder	16	2.02	−0.03
7	Alder	55/20*	1.45	0.09
9	Alder	21	NA	−1.05
10	Alder	4	NA	2.35
6	Ash	16	NA	−0.23
10	Ash	4	NA	1.62
3	Beech	56	1.01	−0.08
3	Birch	56	1.03	−0.05
5	Birch	14	2.32	1.19
5	Birch	14	1.33	−0.95
6	Poplar	18	0.23	−0.22
9	Poplar	21	NA	−1.47
2	Sycamore	24	0.78	−0.71
8	Sycamore	23	NA	0.11
10	Sycamore	4	NA	0.71
	Coniferous			
2	Hybrid larch	24	1.70	0.01
8	Hybrid larch	23	NA	−0.27
3	Scots pine	56	1.67	−0.19
7	Scots pine	55/20*	1.98	0.54
8	Scots pine	23	NA	−0.19
11	Scots pine	4	NA	−2.95
5	Sitka spruce	13	8.56	3.23
7	Sitka spruce	20	3.22	1.31
9	Sitka spruce	21	NA	0.86
	Eucalypts			
1	*E. gunni*	8	10.89	−1.66
4	*E. gunni*	6	NA	−2.19
1	*E. nitens*	8	8.93	−2.36
4	*E. nitens*	6	NA	−1.41

*Replanted after first harvest in 1990, now in second rotation.

also lower when assessing the 1 m soil profile, though this difference was not significant. There are few temperate studies on *Eucalyptus* species with which to compare but in a global meta-analysis and studies in the tropics (Hawaii and Ethiopia) the estimated average annual changes reported are positive. These studies report changes in soil C following afforestation of 0.057 tC ha^{-1} yr^{-1} in global meta-analysis (Paul *et al.*, 2002) and between 0.03 and 5.88 tC ha^{-1} yr^{-1} in individual studies (Kaye *et al.*, 2000; Lemenih *et al.*, 2004; Lemma *et al.*, 2006).

The lower values reported in this study may be related to the relatively young age of the eucalypt SRF plantations investigated (i.e. 6 and 8 years), which may still be within the period of soil C deficit. Further assessments are required to support the findings from the four 'young' transitions studied here, but currently such sites are not available within the United Kingdom (Leslie *et al.*, 2012). Some species of *Eucalyptus* are also characterized by more recalcitrant litter and higher water use than broadleaved and some coniferous tree species and these factors can lead to slower increases in soil C relative to other SRF species (Paul *et al.*, 2002; Vanguelova & Pitman, 2011). In addition, at Site 1, the use of plastic mulch strips for weed suppression at establishment may have further reduced the quantity and rate at which leaf litter has been incorporated into the humic soil layer.

Broadleaved sites studied covered a wider age range (4 –56 years) and were found to have no overall effect on soil C compared to the control in either the short cores (0–30 cm) or in the deep cores (0–100 cm). This finding is in line with the meta-analysis conducted by Guo & Gifford (2002) on effects of conventional forestry on soil C in which the authors found no effect on soil C of establishing broadleaved tree species on pasture, a similar land use to the majority of control sites in this study. While the average annual rate of change of 0.07 tC ha^{-1} yr^{-1} within the 0-30 cm under broadleaved species matched Guo & Gifford (2002), this included species with both negative and positive values ranging from −1.47 to 2.35 tC ha^{-1} yr^{-1}, suggesting that tree species effects may interact with site-specific factors like soil type.

In contrast to the generally higher soil C found under coniferous SRF species in this study, the meta-analysis of Guo & Gifford (2002), and the Norway Spruce (*Picea abies*) chronosequence reported by Thuille & Schulze (2006) both found that, regardless of plantation age, soil C decreased when coniferous trees were established on pasture. This finding also differs from the loss of soil C assessed in paired-sites under coniferous species compared to low productivity grassland in New Zealand (Hewitt *et al.*, 2012), though here several transitions to Scots pine exhibited small reductions in soil C. The higher soil C levels under coniferous SRF compared to broadleaved SRF is also contrary to a number of published studies in which higher values have been reported under broadleaved species (see Laganière *et al.*, 2010; Vesterdal *et al.*, 2013). However, the positive rate of change of 0.70 tC ha^{-1} yr^{-1} under conifers is similar to other studies of afforestation of agricultural soils (e.g., Post & Kwon, 2000). In particular, increases in soil C under Sitka spruce are also supported by the findings from paired-site approaches in Ireland by Black *et al.* (2009) and Wellock *et al.* (2011). Gurmesa *et al.* (2013) found higher soil C under Norway spruce than under either beech (*Fagus sylvatica*) or oak (*Quercus robur*) in a recent common garden experiment. In their review Vesterdal *et al.* (2013), however, caution that impacts of tree species on soil C is highly variable and appear to be both species- and site-specific and thus contrary results are common.

Changes in bulk density did not appear to have a large bearing on the differences in soil C stocks across transitions or between SRF types; relative differences in soil C between coniferous, broadleaved and eucalypt SRF types were similar using both a fixed-depth and mass-based approach. In a paired-site study of afforested pasture and grasslands, Wellock *et al.* (2011) also found that there were limited overall differences between fixed-depth and mass-based methods at 0–30 cm depth as a whole, but that the mass-based approach became more important when considering the distribution of soil C across individual layers e.g., 0–10 cm, 10–20 cm.

The litter layer was more developed in the coniferous SRF types compared to the other land uses, and therefore differences between the control, broadleaved and coniferous SRF in this study may also in part be due to the additional inclusion of the humic layer (L_H) within the soil samples. Due to both difficulties in standardizing the division of litter and mineral soil layers (Hewitt *et al.*, 2012), and recognition of the importance of the C stored within the litter layer, both Laganière *et al.* (2010) and Vesterdal *et al.* (2013) have expressed the need to include the forest floor in assessments of changes in soil C stock. In this study including the litter layer in the assessment of impacts on C stock led to a similar outcome, with higher levels of C under coniferous species than under the broadleaved species and the other land uses. This finding is also in line with other studies in which the litter layer has been included (Thuille & Schulze, 2006; Laganière *et al.*, 2010; Wellock *et al.*, 2011).

Ecosystem C stocks

Guo *et al.* (2008) suggest that to avoid misleading comparisons all above- and belowground C pools should be included when assessing the effects of land-use transitions on C stocks. In this study, the largest difference between SRF and control plots can be seen when total ecosystem C is assessed in the 15 transitions where aboveground measurements are available. Total ecosystem C stock, when just surface soils to 30 cm are included, being between 1.05 and 2.51 times greater in the SRF than the paired control sites. These total C stocks generally increase from control < eucalypts < broadleaves < coniferous.

The assessment of total ecosystem C stocks does, however, require the consideration of the stability of different C pools. The aboveground C stock will be strongly affected by the harvest cycle, with nearly all biomass removed every 8–20 years. Provided the SRF sites are replanted aboveground C stock will recover, but it is important to note that the fate of the harvest biomass should also be considered in any assessments of the wider sustainability of SRF systems (e.g., Hammond *et al.*, 2011; Rowe *et al.*, 2011). Studies on conventional forestry systems suggest that C within the litter layer may be less stable than in mineral soil (Diochon *et al.*, 2009; Nave *et al.*, 2010; Vesterdal *et al.*, 2013). Nave *et al.* (2010), for example, found in their meta-analysis that harvesting resulted in an average loss of soil carbon of 8 ± 3%, but losses in the litter layer were much higher, at −20% for coniferous/mixed stands and −36% for hardwood. The short rotation length in SRF systems provide maximum yields of biomass, but there may need to be a balance struck between yield and impacts on C storage in the soil and litter layers (Jandl *et al.*, 2007). Due to the limited availability of older SRF sites in this study, only one-second rotation was sampled (Site 7). It is therefore not possible to assess any impacts of harvesting and further evidence from second and subsequent rotations remain a priority.

Soil C at depth

In contrast to the other SRF species, coniferous SRF species were found to have higher levels of soil C compared to the control, not only in the surface layers (0–30 cm) but also at depth of 30–50 cm, suggesting C inputs to the deeper soil layer. Such changes could potentially lead to greater increases in C stock, in addition deeper soil layer are less likely to be negatively affected by modern harvest and restock operations on ex-agricultural land. Further work is required to assess the source and stability of carbon in these deeper soil layers but inputs via root exudates, root turnover and mycorrhizae associated with tree species are likely to play a key role. Furthermore, caution should be exercised as additions of labile C to deeper soil layers can have negative as well as positive impacts on older C stocks. (Fontaine *et al.*, 2007). Nevertheless, these results highlight the importance of sampling below 30 cm when making comparisons between such land uses.

Soil acidification

Short Rotation Forest plantations were found to lower soil pH in the 0–15 cm and 15–30 cm soil layers by 0.37 and 0.23 respectively, though the nature of this relationship varied between sites. A number of studies have also reported reduction in soil pH following afforestation (Berthrong *et al.*, 2009; Rigueiro-Rodríguez *et al.*, 2012). In their global meta-analysis, Berthrong *et al.* (2009) reported a similar mean reduction of 0.3 pH units following afforestation. They noted that these results were mainly driven by the coniferous species within their analysis. Tree-induced changes in pH have been found to be linked to changes in both microbial communities and higher trophic levels (Cesarz *et al.*, 2013; Eissfeller *et al.*, 2013). Such cascading impacts on soil biology may have further implications for decomposition and GHG dynamics.

Wider environmental implications

It is important to highlight that while the work presented here fills an important knowledge gap in relation to the sustainability of SRF as a bioenergy feedstock, environmental impacts other than C sequestration have not been considered. GHG fluxes, particularly N_2O, are likely to be significantly affected by development of SRF on agricultural land and may be affected differently by SRF species. These changes must also therefore be taken into account for GHG mitigation potential under such land uses (Smith *et al.*, 2013). Afforestation is also likely to influence a range of ecosystem services such as water flow and biodiversity. For example, coniferous species within this study which were associated with the greatest increase is C stock that have been reported to support lower levels and fewer species of ground flora compared to pasture or broadleaved tree species (Rigueiro-Rodríguez *et al.*, 2012). While these factors were outside the scope of this work, they also require consideration in relation to the development of a sustainable bioenergy supply chain.

In summary, we have shown that the establishment of SRF on agricultural land in GB conditions is likely to result in either no change in soil C or a small increase, depending on the tree species planted, soil type and biomass yield. Initial losses in soil C are, however, to be expected. Taking into account both above- and belowground C stocks within first rotation SRF plantations, total ecosystem C stocks are likely to be in the region of 1.5 times higher than the alternative land uses. While further research is required on the long-term impacts of SRF on soil biogeochemistry, GHG emissions and other ecosystem services, this paper provides part of an evidence base on the sustainability of SRF plantations for policy makers and other end-users.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Figure S1. Examples of selected soil cores taken to 1 m depth. The 0-15, 15-30, 30-50 and 50-100 cm divisions can been seen on the lower two cores. The upper core is shown prior to subdivision. Site names have been blanked.

Table S1. Additional details of sampling locations used to examine the effects of Short Rotation Forestry on carbon stocks in GB. Site number, Region, Latitude and Longitude are repeated from Table 1 for ease of comparison.

Table S2. Soil bulk density and pH measured at two depths in short cores. ND = no data, as soil pH measured in only single plot. Transition units in bold represent control land use.