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Contrasting changes in soil carbon under first rotation, secondary and historic woodland in England and Wales

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Abstract

This study investigates changes in soil carbon under woodland combining data from the National Soil Inventory of England and Wales with data from the National Inventory of Woodlands and Trees to create a unique dataset with woodland management information at the sites where soil carbon was measured in 1980 and 2003. Three woodland management stages were compared: first rotation (i.e. recently planted on land not previously under woodland and not yet harvested), second rotation (i.e. harvested at least once), and historic woodlands. Woodlands in their first rotation demonstrated a reduction ($p < 0.01$) in topsoil organic carbon content typically losing over 2% per year, whereas no change ($p \geq 0.10$) was observed for the other two woodland types. This large reduction in organic carbon could not be statistically explained by a higher inherent soil carbon, as the mean soil carbon content of the first rotation and second rotation woodlands were not ($p \geq 0.50$) different. The average age of the woodlands under first rotation was 42 years, indicating that the period of significant soil carbon loss could go on for about 40 years after planting.

Keywords

Soil organic carbon Forest management Rotation

1. Introduction

Quantification of changes in soil carbon over time is key to understanding the role of soils as sinks or sources of organic carbon. Previous studies (Bellamy et al., 2005; Kirk and Bellamy, 2010) have found large reductions in the carbon concentration of surface soil layers across England and Wales between 1978 and 2003 and related these losses to land use and management practices. In a similar way, a comparison of soil carbon on arable and grassland areas in a mountainous region in France (Saby et al., 2008) showed a mean loss in topsoil soil organic carbon from 26.7 g kg^{-1} in 1990 to 19.8 g kg^{-1} in 2004. This French study highlighted that the greatest loss occurred from soils with high carbon contents, and the role of changes in land use from permanent grassland to cultivation and increased temperatures during the survey period. In New Zealand (Schipper et al., 2007), significant amounts of soil carbon (2.1 kg m^{-2}) were lost in grassland areas over a period of 13–20 years. In a country-wide survey in China (Xie et al., 2007), soil organic carbon declined in grassland areas (−3.56 Pg over 278 Mha; −1.28 kg m^{-2}) over a period of 20 years (1980–2000) which was attributed to grassland degradation and poor management practices. By contrast, other studies have provided evidence of gains in soil organic carbon. For example, Lettens et al. (2005) reported an increase in soil organic carbon from 1960 to 2000 in Belgium in the upper 30 cm for grassland and forested areas. Kirby et al. (2005) reported an increase in soil organic matter for forests in lowland (15.7% to 16.7%), mineral

(7.8% to 12.9%) and organo-mineral (15.6% to 17.6%) soils for British woodlands between 1971 and 2001. Emmett et al. (2010) also reported an increase in soil carbon concentrations in woodlands in Britain between 1978 and 2007. For Danish croplands (Heidmann et al., 2002), the carbon content has been reported to increase on loamy soils over a period of 10–12 years (1986–87 to 1997–98), with annual changes in carbon stocks at 0–50 cm depth ranging from −0.13 to 0.21 kg m^{-2}. Independent of whether the soil is currently sequestering or releasing carbon to the atmosphere, it is useful to understand whether these changes in soil organic carbon are affected by climate-change and land management. If they are affected by land-management, then there is a possibility to modify practices to increase carbon sequestration to soils.

Kirk and Bellamy (2010) showed that the magnitude and scale of climate-change-induced changes in soil carbon in England and Wales was relatively small when compared to the magnitude of land management-induced changes. Subsequently Smith et al. estimated (Smith et al., 2007) that temperature changes due to climate change could have resulted in at most 10–20% of the carbon loss found in England and Wales (Bellamy et al., 2005). Further studies (Cox et al., 2000; Davidson and Janssens, 2006; Heimann and Reichstein, 2008; Jenkinson et al., 1991) looking at detecting climate-change induced losses of soil carbon have recognized the need to account for the strong effect of land management in the analysis. A review (Smith et al., 2007) of the changes in land management practices occurring during the sampling period of the loss found in England and Wales (Bellamy et al., 2005) identified a number of changes including a decrease in spreading of animal manure (Jenkinson, 1988) due to a steady decline in livestock numbers FAO (2005), more efficient removal of agricultural products from fields (Ewert et al., 2005; Smith et al., 2005; Poulton et al., 1996), deeper ploughing depths due to technological advances, and the legacy of land use change occurring before the survey period (Heimann and Reichstein, 2008; Jenkinson, 1988; FAO, 2005; Ewert et al., 2005; Poulton et al., 1996; Smith et al., 2005). However, there are few existing studies of how forest type and management practice affects soil carbon changes of forest land. Hence, this study aims to quantify changes in soil organic carbon in forests in England and Wales and to determine how any changes relate to rotation stage.

2. Materials and Methods

2.1. Data

Two data sets were analysed: the National Soil Inventory (NSI) of England and Wales (Loveland, 1990) carried out by the National Soils Resources Institute and the National Inventory of Woodlands and Trees (NIWT) from Forest Research (Wright, 1998). The NSI consists of 5662 sites that were sampled for soil carbon (g kg^{-1}) in 1980, 40% of which were re-sampled between 1995 and 2003. The re-sampling was done in three phases: in 1994–1995 for arable and rotational grassland sites, in 1995–1996 for managed permanent grassland sites, and in 2003 for non-agricultural sites including woodland. Soils were sampled on an orthogonal 5 km grid; samples were collected and soil profiles characterised at the intersection of each of these grids. The 5 km grid was offset 1 km north and 1 km east of the origin of the National Grid. Urban areas and water bodies were not sampled. A total of 25 soil cores were taken at 4 m intervals at the intersections of a 20 m grid centred on the 5 km grid sampling site. The auger used to extract each of the samples had a 2.5 cm diameter; samples were taken to a maximum depth of 15 cm having removed surface litter. Depending on the carbon content, samples were either analysed for soil organic carbon (C_{org}) using the Walkley-Black method (Kalembasa and Jenkinson, 1973) or by loss on ignition (LOI) (Bellamy et al., 2005). Those samples analysed by LOI were converted to C_{org} by assuming that C_{org} = 0.5 × LOI. An analysis carried out to detect differences between these two methods showed that there was good agreement and no systematic deviation between the values of C_{org} obtained with the LOI and the Walkley-Black methods

(Bellamy et al., 2005). Land use was also recorded at each of the sampled sites following a broad classification (i.e., horticultural crops, permanent grassland, ley grassland, arable, bog, recreation, rough grazing, lowland heath, scrub, deciduous woodland, coniferous woodland, salt marsh, upland grassland, upland heath, orchard, montane and other).

Forest Research carried out the NIWT which surveyed all woodlands in the UK between 1999 and 2003. A digital map showing all woodland over 2 ha was created from 1:25,000 aerial photographs. Woodland was classified from the aerial photographs into broad forest types: conifer, broadleaved, mixed coppice, coppice with standards, shrub, young trees, ground prepared for new planting, and felled (Wright, 1998). New areas planted subsequent to the date of the aerial photographs were added by reference to the Woodland Grant Scheme information and to Forest Enterprise sources. Mapping was followed by a ground survey of roughly 1% of the woodland area. Woodlands were stratified into three size categories with 1 ha plots then being used for ground sampling. Data collected at each plot included information on forest type, management, date of planting, and rotation. Rotation, in this context is a period of time normally sequential (e.g., first or second rotation) where an even aged stand is planted or regenerated, matures and is then felled. First rotation woodland refers to trees planted on land previously not used for woodland (the previous land use is not defined). Second rotation woodland was defined as woodland that has been felled and re-established at least once in its history. The third class was “historic woodland” which was used to indicate both ancient woodland and long-term woodland. Ancient woodlands are areas that have persisted since 1600 in England and Wales, and 1750 in Scotland (Forestry Commission, 2017). Long-term woodland refers to woodland that was established decades ago but it may include trees planted within the stand in more recent years.

Spatially co-located NSI and NIWT sites were selected using Geographical Information Systems to investigate whether the losses in carbon exhibited in the NSI data could be related to management of the woodlands. NSI and NIWT sites were considered to be co-located in space when the NSI point fell within a woodland polygon. Only those sites classified as land cover “woodland” (i.e., coniferous and deciduous) in the second sampling of the NSI were considered for analysis, as these would be the only sites that would have been included in the NIWT which was carried out at about the same time (2003) as the NSI resampling.

2.2. Statistical methodology

The rate of change of organic carbon (ΔC_{org} in g (kg yr)$^{-1}$) between the first (C_{org1} in g kg^{-1}) and second (C_{org2} in g kg^{-1}) NSI surveys was calculated as shown in Eq. (1):

$$\Delta C_{org} = \frac{C_{org_2} - C_{org_1}}{t} \tag{1}$$

where t is the time (years) between the first and second NSI surveys at a particular location. The sampling dates were available for both the first and second sampling for the NSI so the time interval between samplings could be determined to the nearest day. The mean organic carbon (C_{org} in g kg^{-1}) at each NSI site during the sampled period was estimated as shown in Eq. (2).

$$\overline{C_{org}} = \frac{C_{org_1} + C_{org_2}}{2} \tag{2}$$

The analysis of the NSI showed that the rate of reduction in the organic carbon concentration in the surface layer of soils across England and Wales increased linearly with increasing soil carbon content for the whole country and across all land uses; hence losses were greatest in soils with large carbon contents while soils with low carbon contents tended to gain carbon. That carbon loss is a function of soil carbon is not surprising, because at larger quantities of soil carbon there is more carbon to lose.

It was intended therefore to use the organic carbon as a covariate in an Analysis of Covariance (ANCOVA) (Sokal and Rohlf, 1995) to investigate if differences in rate of change in organic carbon were due to the type of woodland (i.e., coniferous and deciduous), as classified by the NSI land use classification. ANCOVA relates values of the dependent variable (i.e., the rate of change of carbon) at each level of a fixed effect (e.g., coniferous or deciduous woodland) allowing for the variation in a continuous variable (i.e., mean of the organic carbon measured at the first and second sampling). However, the model fitted within the ANCOVA analysis is valid only if several assumptions about the data hold. These are that (i) the covariate is normally distributed, (ii) the dependent variable is normally distributed within the groups of the fixed effects, (iii) the variance within the groups is constant and (iv) the slopes of the regressions lines within each group are homogeneous. These assumptions were tested for the dataset analysed through the assessment of descriptive statistics (skewness, kurtosis, mean, median and standard deviation), residual plots, data normality and outliers (Sokal and Rohlf, 1995). Data transformations were carried out and extreme values excluded where necessary. Some of these assumptions were not valid when investigating if differences in rate of change in organic carbon were due to the type of woodland. The Mann-Whitney U test (Lehmann, 1975), a nonparametric alternative to the t-test for independent samples, was used in this instance. This test does not allow for covariates to be included in the analysis but does not require the assumptions of normality or homogeneity of variances.

Both parametric and nonparametric tests were combined in the analysis. The data from the co-located sites from the NSI and the NIWT were used to investigate whether a significant loss of carbon was observed within each rotation class: a t-test was carried out to compare each rotation class with the null hypothesis that the rate of change of carbon in that class equals 0. A t-test was also carried out to assess whether the average amounts of carbon varied with rotation and to investigate the time between planting and survey of the different rotation classes. Normality of the data was assessed within the individual classes and extreme values removed if necessary.

3. Results and discussion

3.1. Results

A total of 210 re-sampled NSI points were under woodland at the second sampling; 106 and 104 under coniferous and deciduous/mixed woodland, respectively (Table 1 and Fig. 1). The mean time interval between sampling for the NSI sites identified was 21.7 ± 0.14 years. From these 210 NSI records, 75 points fell within woodland plots with forest management information, but only 69 (58 under coniferous and 11 under deciduous woodland) had information on both organic carbon change and forestry management (Table 2 and Fig. 1). Management information was not available for all the NIWT points. Hence, those variables with partial records or with only a few sites per sub-class recorded were not considered for analysis; only "rotation" had enough records to be considered for further analysis.

Table 1 summarises the descriptive statistics for the ΔC_{org} in coniferous and deciduous woodlands. Overall, soils under woodland showed a decrease in organic carbon (-0.82 ± 0.25 g (kg yr)$^{-1}$). Fig. 2 shows box-plots of the NSI data for the two groups indicating the large variation in the data and their non-normality (Table 1), which was maintained even on transformation of the data. Although the mean ΔC_{org} for NSI sites under coniferous woodland (−1.09 ± 0.42 g (kg yr)$^{-1}$) was greater than that for sites under deciduous/mixed woodland (–0.53 ± 0.26 g (kg yr)$^{-1}$), this difference was not statistically significant ($p > 0.05$). The nonparametric Mann-Whitney U test was used due to the non-normality of the data. Results showed that there were no significant differences (p–value = 0.86) in the rates of change of carbon between the two types of woodland.

Table 1 Descriptive statistics of rate of change in organic carbon (ΔC_{org} in g (kg yr)$^{-1}$) in coniferous and deciduous woodlands for the NSI sites; N is the number of sites, SD is standard deviation, SE is standard error, Sk is skewness and K is kurtosis. C_{org} in g kg^{-1} is the mean carbon content in the soil.

Woodland	ΔC_{org}									$\overline{C_{org}}$
type	N	Mean	Median	Min	Max	SD	SE	Sk	K	Mean
Coniferous	106	-1.09	-0.199	-18.28	18.25	4.30	0.42	-0.72	7.09	99.64
Deciduous	104	-0.53	-0.07	-12.06	8.37	2.61	0.26	-1.38	5.90	62.43
All woodland	210	-0.82	-0.11	-18.28	18.25	3.57	0.25	-1.00	8.85	81.21

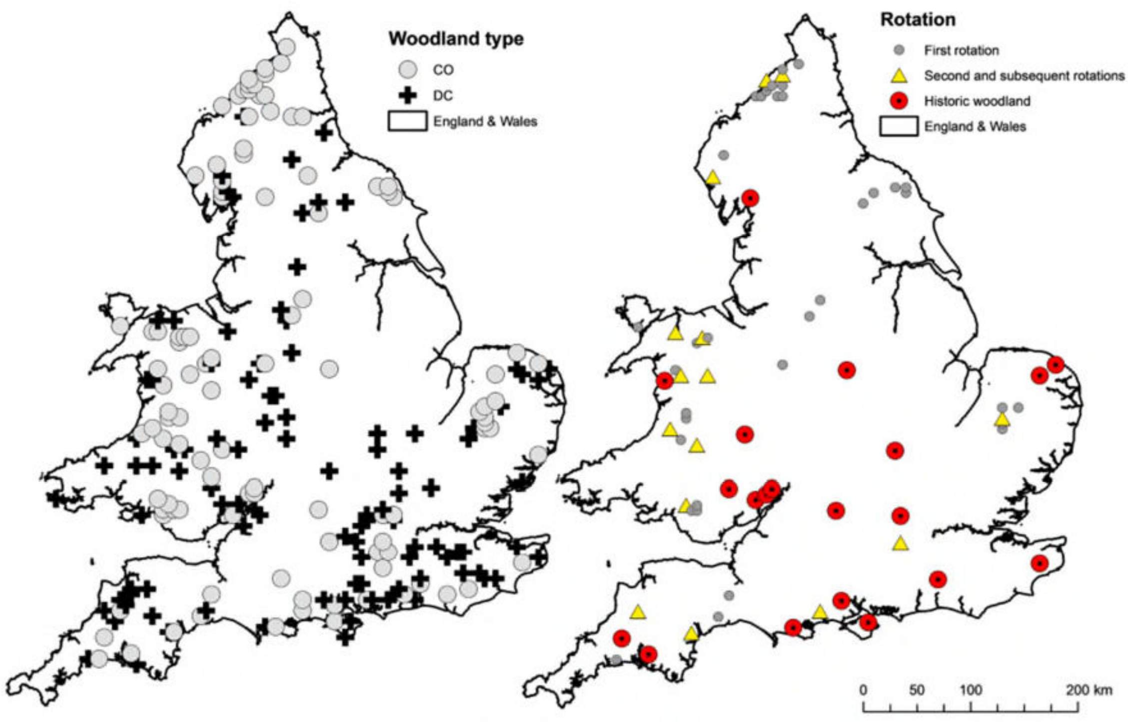

Fig. 1. Map of England and Wales showing (left) the National Soil Inventory sites under woodlands at the second sampling classified as deciduous (DC) and coniferous (CO) woodlands; (right) co-located sites with management information under first rotation, second rotation and historic woodland.

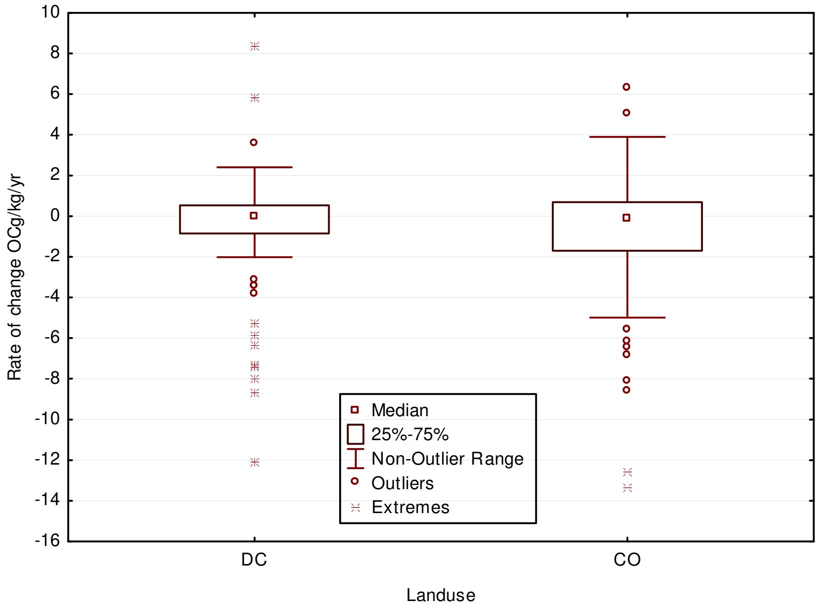

Fig. 2. Box-plots for the National Soil Inventory sites classified as woodland at the second sampling grouped by woodland type (DC = deciduous, CO = coniferous).

Table 2 shows the descriptive statistics of the co-located data for each of the rotation classes. All except one of the sites within the first and second rotations were classified as coniferous woodland within the NSI (Fig. 1). The rates of change were normally distributed within the groups on removal of three outliers within the first rotation class, as was mean carbon once it had been log transformed. All tests were carried out on the data excluding the three outliers and using the transformed means. Results for the t-test, carried out to assess the mean rate of change for each rotation class compared to a zero change in carbon, showed that for both the second rotation (−1.22 g $(kg\ yr)^{-1}$) and the historic woodland (0.01 g $(kg\ yr)^{-1}$) there was no significant change in organic carbon levels in the surface layers (p-value = 0.10; p-value = 0.99 respectively). By contrast those sites under the first rotation showed a significant rate of loss of carbon (–2.71 g $(kg\ yr)^{-1}$, p-value < 0.01) (Fig. 3). As one would expect sites with a higher inherent, or pre-existing, carbon content to lose carbon more quickly, the mean soil organic carbon was compared between the first and second rotations. No significant difference was found (t-test on log C_{org} , p-value = 0.72) indicating that the significant loss in carbon shown by first rotation could not be statistically related to those woodlands being on higher carbon soils than the second rotation.

The C_{org} in the soil was highly variable for those sites within each woodland class (Table 3). For sites under first rotation, C_{org} ranged between 1.45 and 468.4 g kg^{-1}, whereas for sites under second rotation the values ranged from 15.60 to 296.37 g kg^{-1}. Historic woodland C_{org} content presented a narrower range (17.75 to 133.40 g kg^{-1}) than that observed for the other groups.

Table 2 Descriptive statistics of rotation classes; N for the number of sites, ΔC_{org} in g (kg yr)$^{-1}$ for the rate of change in organic carbon, C_{org} in g kg^{-1} is the average carbon content in the soil calculated as in Eq. (2), SD is standard deviation, SE is standard error, Sk is skewness, K is kurtosis and Age is the age of woodland (years).

		ΔC_{org}								$\overline{C_{org}}$	Age		
Rotation Class	N	Mean	Median	Min	Max	SD	SE	Sk	K	Mean	N	Mean	SE
First rotation on formerly pasture land	34	-2.94	-1.43	-17.80	18.25	6.29	1.08	0.30	3.75	135.59	26	42.15	1.87
First rotation on formerly pasture land (excluding outliers)	31	-2.71	-1.16	-12.59	2.37	3.85	0.69	-1.02	0.11	124.28	23	41.78	2.10
Second and subsequent rotations	15	-1.22	-0.81	-6.24	3.89	2.71	0.70	-0.03	-0.27	104.34	10	11.20	2.48
Historic woodland	20	0.01	0.20	-8.71	5.83	2.83	0.63	-0.92	5.11	53.96	17	51.65	13.44
All sites	69	-1.72	-0.45	-17.81	18.25	4.96	0.60	-0.23	5.17	105.14	53	39.35	4.76
All sites (excluding outliers)	66	-1.55	-0.43	-12.59	5.83	3.49	0.43	-1.01	1.33	98.44	50	39.02	5.05

Table 3 Descriptive statistics of the mean carbon content in the soil (C_{org}) in g kg^{-1} for each rotation class; N is the number of sites, SD is standard deviation, SE is standard error, Sk is skewness and K is kurtosis.

Rotation class	$\overline{C_{org}}$								
	N	Mean	Median	Min	Max	SD	SE	Sk	K
First rotation on formerly pasture land (excluding outliers)	31	124.28	94.75	1.45	468.40	112.89	20.27	1.38	1.63
Second and subsequent rotations	15	104.34	72.80	15.6	296.37	83.91	21.66	1.50	1.66
Historic woodland	20	53.96	46.10	17.75	133.40	34.66	7.75	1.27	0.72

Table 2 also reports the ages of the woodlands recorded in the NIWT calculated by taking the date of planting from the date of survey. These data were not available for all those sites with rotation information and change in carbon data. The woodlands in their first rotation had a significantly higher age than those in their second rotation (p-value < 0.001). The mean age of the woodlands under first rotation was 42 years, which indicates that most of the NSI records considered should have been classified as woodland on both the first and second sampling. Of the 34 sites in the first rotation class only four where not classed as under woodland at the first NSI sampling. Three of these sites had NIWT planting dates prior to the first NSI sampling so the land use classification at the time of original NSI sampling must have been incorrect – or the trees had not grown enough to be distinguished from the "upland grazing" land use class which was assigned. There was no relationship between age and rate of change of carbon observed directly using the 23 sites where both values had been observed.

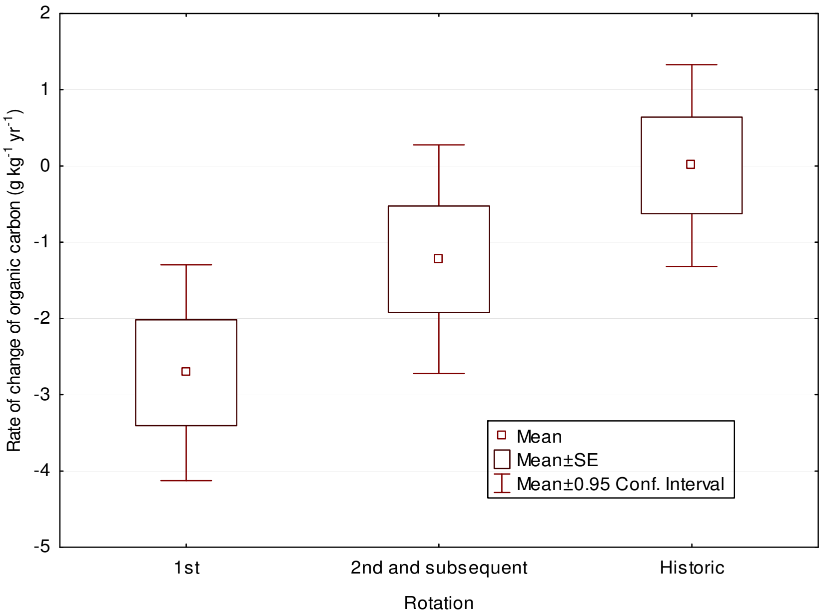

Fig. 3. Box-plots of the rate of change of soil organic carbon for the National Soil Inventory sites with management information grouped by woodland management type (1st = first rotation, 2nd and subsequent rotation, and historic) as classified by the National Inventory of Woodlands and Trees.

3.2. Discussion

Using a subset of the NSI resampled data and information from the NIWT a significant reduction in organic carbon content of the surface soil was detected in land under first rotation woodland – the other woodland categories considered (second rotation and historic woodland) showed no reduction ($p > 0.05$). There was not enough quantitative evidence in this analysis to reject the null hypotheses that the loss in organic carbon content for second rotation and historic woodland rotation classes were equal to 0. The trends observed align with earlier work by Wilson et al. (1997). In Wilson et al. (1997), the characteristics of soils in three ancient woods in UK were compared with woods planted more recently. The quantity of organic matter and carbon found in recent woodland soils, where cultivation preceded establishment, was less than the quantity observed in ancient woodlands. In a further study, Wilson and Puri (2001) compared the carbon content of ancient and semi-natural Scots pine forest with adjacent moorland and found that that forest soils had accumulated significantly more carbon than moorland soils and were a significant carbon sink (total quantities of soil carbon ranging from 21.5 kg m^{-2} (15 cm depth) to 46.1 kg m^{-2} (50 cm depth)). The combined results for these studies seem to suggest that soils under second rotation and historic woodlands could be a more stable store of carbon than other land-uses. Further research should be conducted to ascertain the validity of this hypothesis.

The organic carbon content of the surface layers of the soil is primarily determined by the balance between carbon input (e.g., litter fall and rhizodeposition) and the release of carbon by decomposition. The decomposition rate is a function of the quality of the carbon compounds (e.g., lignin compounds are more difficult to digest by the microbes and therefore have longer resilience than cellulose), site conditions (e.g., temperature, humidity) and soil properties (e.g., soil structure, pH) (Davidson and Janssens, 2006; Giardina and Ryan, 2000; Knorr et al., 2005; Liski et al., 1999; Trumbore et al., 1996).

A review (Jandl et al., 2007) of European studies looking at the effect of tree species indicated that coniferous trees, with shallow roots, tended to accumulate more carbon in the organic litter layer but less in the underlying mineral soil than deciduous species. De Vries et al (2003) reported in a European study that the median soil carbon pools below pine and for woodlands (5.7-13.0 kg m^{-2}) were similar to those below oak and beech woodlands (8.7-14.4 kg m^{-2}). Similarly other studies (Vesterdal and Raulund-Rasmussen, 1998) have related greater carbon storage in the litter layers of coniferous forest to lower decay rates than in deciduous woodland. The level of soil organic content in woodland is also dependent on site conditions and, in the case of England and Wales, conifer forests were historically planted on “less productive” land (e.g., the uplands of Wales and Northern England) which tend to have higher soil organic carbon contents.

This study has shown that, on average, the surface soil layers of coniferous forests within their first rotation lost soil organic carbon at the rate of about 2.7 g $(kg\ yr)^{-1}$, which is a relative rate of about −2.2% per year (relative to the mean soil carbon content of 124.3 g kg^{-1}; Table 3). However it should be noted that the mean content is derived from a sample where the value of C_{org} for first rotation woodland ranged from 1.45 to 468.5 g kg^{-1} . Fig. 4 depicts the relative rate of change of soil carbon for all first rotation sites, where the relative rate of change has been calculated relative to the soil carbon content of each site rather than 124.3 g kg^{-1} .

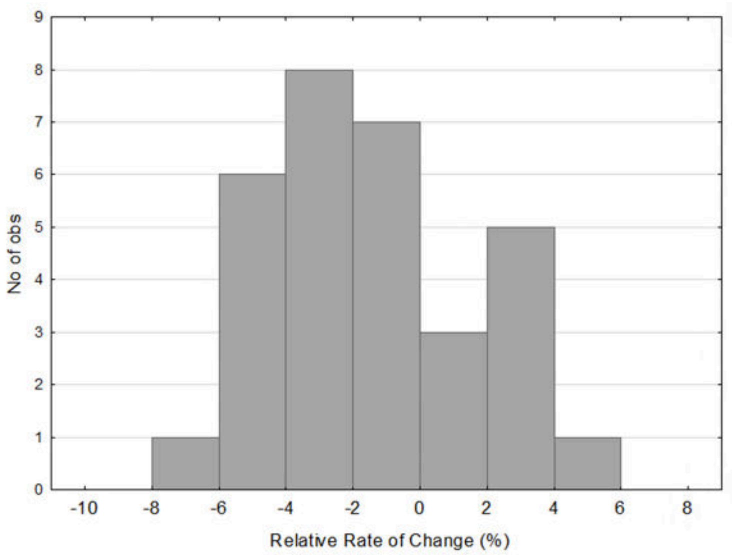

Fig. 4. Histogram for the relative rate of change of carbon (%) (relative to the specific soil carbon content of each forest) across all sites (n = 31) under first rotation. Note that the average relative rate of change (relative to the existing average soil carbon content of 124.3 g kg^{-1} as reported in Table 2) is −2.2% per year.

In a previous study, Kirk and Bellamy (2010) related the majority of the loss of soil carbon found across all land uses in England and Wales (Bellamy et al., 2005) to the residual effects of land use change that occurred before the original sampling. For example, arable land that was previously grassland or forest would continue to lose carbon for many years after the original conversion of land use. The land use before planting is not known for this study, but as stated above conifer forests tend to be planted in England and Wales on less productive land so the land cover was more likely to be pasture than arable crops. The net effect on organic carbon levels near the soil surface of changing land use from pasture to woodland in England and Wales is still a focus of debate. A global meta-analysis (Guo and Gifford, 2002) indicated higher soil carbon levels under pasture than woodland, but this effect could be confounded by site and environmental effects. More recent measurements on long-term trials and experiments in the UK have demonstrated a decrease in soil organic carbon from tree planting on grassland (Upson et al., 2016), and some have shown the lack of a statistically significant effect (Beckert et al., 2016; Fornara et al., 2018). On mineral and organo-mineral soils in the UK, WCA Environment Ltd (2009) reported a net loss in soil carbon from permanent pasture to forest, but within the UK Greenhouse Gas Inventory, Brown et al. (2020) assume an increase in the equilibrium soil carbon density to 1 m depth with a move from grassland to forest land. A recent study using the Woodland Creation and Ecological Networks in the UK (Ashwood et al., 2019) found statistically similar levels of soil carbon under pasture and woodland, but there was a non-significant trend of lower carbon levels under young woodland which then turned to increasing carbon levels with time so that the largest carbon levels occurred under mature woodland.

An initial decline in soil carbon below recently planted woodlands on grassland has been related to the process of tree planting disturbing the soil and leading to the mineralization of soil organic matter. Furthermore, litter fall can be low whilst trees are still developing their canopy, and the turnover of fine roots by recently established trees will be less than that found in a well-established pasture (Upson et al., 2016). Changes in the aeration of the soil due to tree planting may also be important (Upson et al., 2016), and a substantial movement of carbon from the litter layer to the mineral soil may only occur after several decades (Jenkinson et al., 1991; Paul et al., 2003). Some of the soil C may be lost due to leaching as dissolved organic carbon (DOC), especially because of the disturbance involved with the site preparation (Johnson et al., 1995). It is also possible that the carbon could be redistributed to deeper depths (Johnson et al., 1995) but we do not have the information required to investigate this further.

Modelling of the effect of trees on soil carbon across seven sites in Australia (mean annual temperature = 10.2°C to 20.4°C) has been used (Paul et al., 2003) to show that in the first 10 years after planting the soil carbon reduces by on average 1.7% per year and is then predicted to increase by about 0.82% per year from then on. Assuming a nominal soil carbon content of 100 g at planting, a decrease of 17% in the first ten years followed by an addition of 8% in the next ten years would be equivalent to a decrease of 0.45% per year for the 20 year period. The model (Paul et al., 2003) was devised to represent the top 30 cm of soil whereas the data used in this study were measured in the top 15 cm of soil. In addition, conditions and management are different between Australia and the UK so a direct comparison with the results here is difficult.

The observed loss of soil carbon due to afforestation has implications for the estimation of greenhouse gas (GHG) emissions from land use, land use change and forestry (LULUCF) at ecosystem level. In the UK the effect of afforestation is assumed to be a sink for GHGs (Ostle et al., 2009). The findings of the data reported here suggest that these assumptions need to be re-examined to ensure the loss of carbon from the soil is correctly represented in the figures and natural capital accountability is carried out accurately. The eventual increase in surface soil carbon levels as woodland is extended to a second or subsequent rotation is supported by the observation that there was not a significant decrease in the soil C of such woodlands. Whilst harvesting may result in some organic carbon loss, residues from the harvesting (e.g. roots and brash) could offset the decomposition due to the disturbance. The historic woodlands (which include ancient and long term woodlands) also showed no change in the carbon content of the soil surface.

A confounding factor in the analysis of surface soil carbon levels under woodland is that the level of the soil surface can change. For example, under full canopy woodlands a deep litter layer can be created which in time may be recognized as soil. Hence it is theoretically possible for the carbon level in the surface layer to decrease whilst the overall carbon content of the soil increases because of the increased depth. An additional factor to consider is that the calculation of the soil carbon content requires the removal of fibrous tissue and roots. If the carbon content of roots and such fibrous matter is included, then it is possible that whilst woodlands may initially reduce the soil carbon content of previous pastureland, the overall stock of carbon may be similar or increase due to the presence of an increased mass of tree roots.

The 20 historic woodlands where data have been collected for this study were measured over approximately the same interval of time as those in a Chinese study (Zhou et al., 2006) where large increases in soil carbon concentration were observed. The soil depth considered was similar (20 cm compared to 15 cm in this study) but the samples were all taken from a single 200 m × 400 m plot. In our study there were sites where large increases in soil C were observed – the maximum increase in carbon concentration measured was 5.83 g C per kg of soil per year (Table 2) over 10 times as much as that observed at the Chinese site (0.35 g C per kg of soil per year) demonstrating the limitations of using a single site to draw general conclusions. However, an extensive review of the capacity of old-growth forests to act as a carbon sink (Luyssaert et al., 2008) covering over 500 plots demonstrate that, taken as a whole, oldgrowth forests can be a carbon sink – the above ground increase in carbon dominating any small increases or no change in soil C which is consistent with our results in terms of soil C. The average soil C in the historic woodlands was lower than the other rotation classes. Further studies should investigate the reasons for this: potential causes could be that the historic woodland includes coniferous, mixed and deciduous woodland whereas the majority of the first and second rotations are coniferous; or the historic woodlands had established in areas with inherently lower soil carbon contents. Data available for this study did not allow us to address this question.

This study looked at those NSI meta-points that were classified as woodland during the re-sampling of soil carbon. However, it is possible that some NSI records located in woodland areas were excluded from the analysis due to the broad scope of the land use classes used by the NSI. For example, the "recreation area" in the NSI classification could well include woodland areas. Future NSI and NIWT surveys should ensure that a similar land use typology is used across institutions to enable a more comprehensive statistical analysis. The NSI land use classification also showed some inconsistencies with the NIWT classification. Some of these inconsistencies are probably the result of the NSI and NIWT data not being co-located in time; the date of assessment of the NIWT records was between 1999 and 2003 whereas the second sampling of the NSI woodland sites took place in 2003. In this project, it was assumed that woodlands which were in their first rotation were sites where trees had been planted on ground not previously under woodland at some point before the second sampling of the NSI.

4. Conclusions

Analysis of data sampled over a period of about 20 years at NSI sites within forest plots has shown that at sites which are in their first rotation, which we assume were typically planted on grassland, had lower levels organic carbon in the surface soil layers in the second measurement than the first. By contrast the change in secondary and historic woodland was not significant. Such results would support modelling studies that predict an initial decline in soil carbon (which excludes carbon stored as roots or leaf litter) when planting trees on pasture. Because the mean age of the first rotation stands was 40 years, the observed negative effect of tree planting on the organic carbon level of the soil surface could take longer than the 30 years to reach equilibrium suggested by a modelling study (Paul et al., 2003).

Credit authorship contribution statement

Mónica Rivas Casado: Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Data curation, Writing – original draft, Writing – review & editing. Patricia Bellamy: Conceptualization, Writing – original draft, Writing – review & editing, Resources, Supervision, Project administration, Funding acquisition. Paul Leinster: Writing – review & editing. Paul J. Burgess: Writing – review & editing.

Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Monica Rivas Casado reports financial support was provided by DEFRA.

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References

Ashwood, F., Watts, K., Park, K., Fuentes-Montemayor, E., Benham, S., Vanguelova, E.I., 2019. Woodland restoration on agricultural land: long-term impacts on soil quality. Restor. Ecol. https://doi.org/10.1111/rec.13003Citati.

Beckert, Marvin R., Smith, Pete, Lilly, Allan, Chapman, Stephen J., 2016. Soil and tree biomass carbon sequestration potential of silvopastoral and woodland-pasture systems in North East Scotland. Agrofor. Syst. 90 (3), 371–383.

Bellamy, P.H., Loveland, P.J., Bradley, R.I., Lark, R.M., Kirk, G.J.D., 2005. Carbon losses from all soils across England and Wales 1978–2003. Nature 437, 245–248.

Brown, P., Cardenas, L., Choudrie, S., Jones, L., Karagianni, E., MacCarthy, J., Passant, N., Richmond, B., Smith, H., Thistlethwaite, G., Thomson, A., Turtle, L., Wakeling, D., 2020. Annexes to UK Greenhouse Gas Inventory, 1990 to 2018 Annual Report for Submission under the Framework Convention on Climate Change, https://ukair.defra.gov.uk/assets/documents/reports/cat09/2004231037_ukghgi-9018_Annex_v02-00.pdf

Cox, P.M., Betts, R.A., Jones, C.D., Spall, S.A., Totterdell, I.J., 2000. Acceleration of global warming due to carbon-cycle feedbacks in a coupled climate model. Nature 408, 184–187.

Davidson, E.A., Janssens, I.A., 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature 440, 165–173.

de Vries, W., Reinds, G.J., Posch, M., Sanz, M., Krause, G., Calatyud, V., Dupouey, J., Sterba, H., Gundersen, P., Voogd, J., Vel, E., 2003. Intensive Monitoring of Forest Ecosystems in Europe. EC. UN/ECE, Brussels, Technical Report.

Emmett, B.A., Reynolds, B., Chamberlain, P.M., Rowe, E., Spurgeon, D., Brittain, S.A., Frogbrook, Z., Hughes, S., Lawlor, A.J., Poskitt, J., Potter, E., Robinson, D.A., Scott, A., Wood, C., Woods, C., 2010. Centre for Ecology & Hydrology. (Natural Environment Research Council). February 2010. http://www.countrysidesurvey.org.uk/sites/default/files/CS_UK_2007_TR9-revised%20%20Soils%20Report.pdf.

Ewert, F., Rounsevell, M.D.A., Reginster, I., Metzger, M., Leemans, R., 2005. Future scenarios of European agricultural land use I: estimating changes in crop productivity. Agric. Ecosyst. Environ. 107, 101–116. FAO, 2005. Global Livestock Production and Health Atlas (GLiPHA), http://www.fao.org/ag/aga/glipha/index.jsp (accessed on 12 September 2005).

Forestry Commission, 2017. The UK Forestry Standard, Edinburgh 2017. The UK Forestry Standard (publishing.service.gov.uk). Forestry Commission.

Fornara, D.A., Olave, R., Burgess, P.J., Delmer, A., Upson, M., McAdam, J., 2018. Land use change and soil carbon pools: evidence from a long-term silvopastoral experiment. Agrofor. Syst. 92, 1035–1046.

Giardina, C.P., Ryan, M.G., 2000. Evidence that decomposition rates of organic carbon in mineral soil do not vary with temperature. Nature 404, 858–1561.

Guo, L.B., Gifford, R.M., 2002. Soil carbon stocks and land use change: a meta analysis. Glob. Change Biol. 8 (4), 345–360.

Heidmann, T., Christensen, B.T., Olesen, S.E., 2002. Changes in soil C and N content in different cropping systems and soil types. Report 81. In: Greenhouse Gas Inventories for Agriculture in the Nordic Countries, Petersen, S.O.; Olesen, J.E., (Eds.), Ministry of Food, Agriculture and Fisheries, Danish Institute of Agricultural Sciences, Foulum, Denmark, pp. 77–86.

Heimann, M., Reichstein, M., 2008. Terrestrial ecosystem carbon dynamics and climate feedbacks. Nature 451 (7176), 289–292.

Jandl, R., Lindner, M., Vesterdal, L., Bauwens, B., Baritz, R., Hagedorn, F., Johnson, D. W., Minkkinen, K., Byrne, K.A., 2007. How strongly can forest management influence soul carbon sequestration? Geoderma 137, 253–268.

Jenkinson, D.S., 1988. Soil organic matter and its dynamics. In: Wild, A. (Ed.), Russell's Soil Conditions and Plant Growth, 11th ed. Longman, Harlow, pp. 564–607. https://repository.rothamsted.ac.uk/item/860yq/soil-organic-matter-and-its-dynamics.

Jenkinson, D.S., Adams, D.E., Wild, A., 1991. Model estimates of CO2 emissions from soil in response to global warming. Nature 351, 304–306.

Johnson, C.E., Driscoll, C.T., Fahey, T.J., Siccama, T.G., Hughes, J.W., 1995. Carbon dynamics following clear-cutting of a northern hardwood forest. In: McFee, W.W., Kelly, J.M., (Eds.) Carbon forms and functions in forest soils, Soil Science Society of America, Inc., Madison, Wisconsin USA, pp 463-488.

Kalembasa, S.J., Jenkinson, D.S., 1973. A comparative study of titrimetric and gravimetric methods for the determination of organic carbon in soil. J. Sci. Food Agric. 24, 1085–1090.

Kirby, K.J., Smart, S.M., Black, H.I.J., Bunce, R.G.H., Corney, P.M., Smithers, R.J., 2005. Long term ecological change in British woodland (1971–2001). English Nature Research Report 653 English Nature, Peterborough, UK, pp. 1–139.

Kirk, G.J.D., Bellamy, P.H., 2010. Analysis of changes in organic carbon in mineral soils across England and Wales using a simple single-pool model. Eur. J. Soil Sci. 61 (3), 406–411.

Knorr, W., Prentice, I., House, J., Holland, E., 2005. Long-term sensitivity of soil carbon turnover to warming. Nature 433, 298–301.

Lehmann, E.L., 1975. Nonparametrics: Statistical Methods Based On Ranks Holden-Day: Oakland, CA.

Lettens, S., van Orshoven, J., van Wesemael, B., Muys, B., Perrin, D., 2005. Soil organic carbon changes in landscape units of Belgium between 1960 and 2000 with reference to 1990. Glob. Change Biol. 11, 2128–2140.

Liski, J., Ilvesniemi, H., Mäkelä, A., Westman, C.J., 1999. CO2 emissions from soil in response to climatic warming are overestimated—the decomposition of old soil organic matter is tolerant to temperature. Ambio 28, 171–174.

Loveland, P.J., 1990. The National Soil Inventory: Survey design and sampling strategies. In: Lieth, H., Markert, B. (Eds.), Element Concentration Cadasters in Ecosystems; Verlagsgesellschaft, Weinheim, Germany, pp.73–80.

Luyssaert, S., Schulze, E.D., Borner, A., Knohl, A., Hessenmoller, D., Law, B., Ciais, P., Grace, J., 2008. Old-growth forests as global carbon sinks. Nature 455, 213–215.

Ostle, N.J., Levy, P.E., Evans, C.D., Smith, P., 2009. UK land use and soil carbon sequestration. Land Use Policy 26, 274–1273.

Paul, K.I., Polglase, P.J., Richards, G.P., 2003. Predicted change in soil carbon following afforestation or reforestation, and analysis of controlling factors by linking a C accounting model (CAMFor) to models of forest growth (3PG), litter decomposition (GENDEC) and soil C turnover (RothC). For. Ecol. Manage. 177, 485–501.

Poulton, P.R., 1996. The Park Grass experiment 1856–1995. NATO ASI Series I Volume 38. In: Evaluation of Soil Organic Matter Models Using Existing Long-Term Datasets; Powlson D.S., Smith P., Smith J.U., Eds.; Springer-Verlag, Heidelberg, pp 377–384.

Saby, N.P.A., Arrouays, D., Antoni, V., Lemercier, B., Follain, S., Walter, C., Schvartz, C., 2008. Changes in soil organic carbon in a mountainous French region, 1990–2004. Soil Use Manag. 24, 254–262.

Schipper, L.A., Baisden, W.T., Parfitt, R.L., Ross, C., Claydon, J.J., Arnold, G., 2007. Large losses of soil C and N from soil profiles under pasture in New Zealand during the past 20 years. Glob. Change Biol. 13, 1138–1144.

Smith, P., Chapman, S.J., Scott, A., Black, H.I.J., Wattenbach, M., Milne, R., Campbell, C. D., Lly, A.L., Ostle, N., Levy, P.E., Lumsdon, D.G., Millard, P., Towers, W., Zaehle, S., Smith, J., 2007. Climate change cannot be entirely responsible for soil carbon loss observed in England and Wales, 1978–2003. Glob. Change Biol. 13 (12), 2605–2609.

Smith, J.U., Smith, P., Wattenbach, M., Zaehle, S., Hiederer, R., Jones, R.J.A., Montanarella, L., Rounsevell, M.D.A., Reginster, I., Ewert, F., 2005. Projected changes in mineral soil carbon of European croplands and grasslands, 1990–2080. Glob. Change Biol. 11, 2141–2152.

Sokal, R.R., Rohlf, F.J., 1995. Biometry: the principles and practice of statistics in biological research. 3rd edition. W. H. Freeman and Co.: New York, 887 pp.

Trumbore, S.E., Chadwick, O.A., Amundson, R., 1996. Rapid exchange between soil carbon and atmospheric carbon dioxide driven by temperature change. Science 272, 393–396.

Upson, M.A., Burgess, P.J., Morison, J.I.L., 2016. Soil carbon changes after establishing woodland and agroforestry trees in a grazed pasture. Geoderma 283, 10–20.

Vesterdal, Lars, Raulund-Rasmussen, Karsten, 1998. Forest floor chemistry under seven tree species along a soil fertility gradient. Can. J. For. Res. 28 (11), 1636–1647.

WCA Environment Limited, 2009. Defra Project SP0567: Assembling UK Wide Data on Soil Carbon (and Greenhouse Gas Fluxes) in the Content of Land Management. Final Report to Defra, UK. http://randd.defra.gov.uk/Document.aspx?Document=SP0567_995 3_FRP.pdf

Wilson, B.R., Moffat, A.J., Nortcliffs, S., 1997. The nature of three ancient woodland soils in southern England. J. Biogeogr. 24 (5), 633–646.

Wilson, B.R., Puri, G., 2001. A comparison of pinewood and moorland soils in the Abernethy Forest Reserve, Scotland. Glob. Ecol. Biogeogr. 10 (3), 291–303.

Wright, D., 1998. The National Inventory of Woodland and Trees. Information Note 8. Forestry Commission, Edinburgh.

Xie, Z., Zhu, J., Liu, G., Cadisch, G., Hasegawa, T., Chen, C., Sun, H., Tang, H., Zeng, Q., 2007. Soil organic carbon stocks in China and changes from 1980s to 2000s. Glob. Change Biol. 13, 1989–2007.

Zhou, G., Liu, S., Li, Z., Zhang, D., Tang, X., Zhou, C., Yan, J., Mo, J., 2006. Old-growth forests can accumulate carbon in soils. Science, 314 (5804), 1417.

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