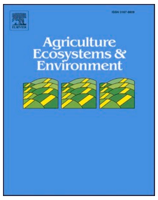
Agriculture, Ecosystems and Environment 382 (2025) 109471

Contents lists available at ScienceDirect

Agriculture, Ecosystems and Environment

journal homepage: www.elsevier.com/locate/agee



Consistent soil organic carbon accumulation under hedges driven by increase in light particulate organic matter

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ARTICLE INFO

Keywords:
Agroforestry
Carbon accounting
Fractionation
Climate change mitigation
POM
MAOM
Sequestration

ABSTRACT

Hedgerow planting is recommended by biodiversity policies and those that promote the inclusion of woody plants in agricultural landscapes to sequester atmospheric carbon into the soil. However, the extent and variability of soil organic carbon (SOC) sequestration under hedges are not known. We measured SOC stock beneath hedges in five pedoclimatic conditions in the UK to quantify the SOC sequestration potential associated with hedgerow planting. We measured SOC stocks in 10 cm intervals in the top 50 cm of soil or to bedrock, comparing 46 hedges of different age classes and their adjacent grassland fields. We assessed how additional SOC stocks and SOC sequestration rates under hedges varied with covariates of climate and soil properties. The mean additional SOC stock under hedges was consistent across pedoclimatic conditions at ~40 Mg C ha^{-1} more than improved grassland fields. On average, SOC stocks beneath hedges were 40 % higher than in adjacent fields at 0–50 cm depth, with older hedges storing greater additional SOC stock at depth than younger hedges. The additional stock was driven by an increase in light particulate organic matter (l-POM), due to increased leaf and root litter inputs under woody vegetation. The mean SOC sequestration rate of mature hedges was 1.5 (1.0–2.0) Mg C ha^{-1} yr^{-1} while the net SOC sequestration rates over time since hedgerow planting declined from 4.2 to 0.2 Mg CO$_2$ km^{-1} yr^{-1} within the first 20 years. Our results will aid future land-use related carbon accounting and inform climate change mitigation practice.

1. Introduction

Recent policy initiatives have placed a strong focus on the use of agricultural soils for atmospheric CO$_2$ removal by adopting practices that increase soil organic carbon (SOC) storage (Minasny et al., 2017; Bossio et al., 2020; Rodrigues et al., 2021; Gelardi et al., 2023). Thus, changes in agricultural land-use, such as the integration of woody species in agricultural landscapes, have been recommended for climate change mitigation and contribute towards meeting the Paris Climate Agreement (IPCC, 2006; Zomer et al., 2016; Terasaki Hart et al., 2023). Hedges – lines of woody plants commonly used to delineate agricultural fields– provide multiple ecosystem services (Baudry et al., 2000; Montgomery et al., 2020; Boinot et al., 2023), including food and shelter for livestock, wildlife, and remnant plant species (Staley et al., 2015; Litza et al., 2022; Biffi et al., 2024), nutrient interception (Holden et al., 2019), and carbon (C) storage both in their biomass and in the soil beneath them (Axe et al., 2017; Biffi et al., 2022, 2023; Drexler et al., 2021, 2023). For these reasons, ambitious hedge planting targets have been set throughout Europe, such as in Germany, Denmark, France, and Ireland (Drexler et al., 2021; Levin et al., 2020; MAA, 2021; Black et al., 2023). In the UK, hedge planting has been widely promoted to meet the UK's net-zero carbon commitments and other environmental targets

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https://doi.org/10.1016/j.agee.2025.109471
Received 12 August 2024; Received in revised form 21 November 2024; Accepted 2 January 2025
Available online 11 January 2025

outlined in the 25 year Environment Plan (DEFRA, 2021; Thomas, 2022; Woodland Trust, 2021; Burgess and Graves, 2022). In particular, the UK Climate Change Committee proposed increasing the hedge network by 40 % by 2050 (Climate Change Committee, 2018) and, more recently, the UK Government announced its goal of planting 72,500 km of hedges throughout England by 2050.

Understanding the drivers of SOC storage under hedges is important, as a variable hedge planting SOC sequestration potential across different pedoclimatic conditions would impact the overall contribution of hedge planting goals towards net-zero and climate change mitigation targets. When planting hedges, the initial SOC stock is likely to affect SOC changes in response to land-use change (Vityakon et al., 2000; Stewart et al., 2008a; Castellano et al., 2015). The initial SOC stock is largely regulated by the interplay of climatic and pedological conditions, and C inputs (Jobbágy and Jackson, 2000; De Deyn et al., 2008; Luo et al., 2017; Vos et al., 2019; Heckman et al., 2022). Temperature and moisture are often regarded as the main driver of SOC stocks, as they govern a large part of SOC variability at both global and regional scales (Meier and Leuschner, 2010; Carvalhais et al., 2014). Soil texture and parent material also play an important role, as the majority of SOC is bound to the fine mineral fractions of silt and clay (Plaza et al., 2013; Castellano et al., 2015), while soil chemical properties, such as pH, influence SOC by affecting soil organic matter (SOM) mineralization rates (Aciego Pietri and Brookes, 2008). When under similar intensively managed land-use, the variability in initial SOC stocks due to C inputs is largely due to potential variability in management practices (e.g. fertiliser input, grazing pressure, number of silage cuts, Eze et al., 2018b). Hedge planting, instead, may increase C inputs quantity and quality due to changes in the fine root and hyphal turnover, exudation, and accumulation of leaf litter (Godbold et al., 2006; Yan et al., 2018) under hedges compared to improved grassland. For instance, residues of woody plants with high lignin content can slow decomposition rates by altering soil microbiota composition (Austin and Ballaré, 2010), increasing SOC compared to herbaceous residues.

The land-use change from grassland to hedge may also have an impact on the SOM fractions among which SOC is distributed, which have different ecological functioning and rates of degradation, thus controlling the SOC stock stability under hedges (Bailey et al., 2019; Lugato et al., 2021; Bramble et al., 2023). An increasing research focus has been placed on labile SOM fractions, such as particulate soil organic matter (POM), which consists of chemically recalcitrant litter C compounds of partially decomposed plant residues (Haddix et al., 2016; Robertson et al., 2019). This fraction can be distinguished into free or light POM (l-POM) and occluded or heavy POM (h-POM) based on the degree of physical occlusion of the SOM particles within soil aggregates, with l-POM being the most readily available to decomposition under suitable environmental conditions. Mineral-associated organic matter (MAOM), instead, is regarded as the most persistent of the three SOM fractions, as it is physiochemically more protected from decomposition, and thus more resistant to land-use changes and climate change (Stewart et al., 2008b; Totsche et al., 2018; Hemingway et al., 2019; Schweizer et al., 2021; Heckman et al., 2022). The amount of POM in a soil is largely dependent on continuous input of litter and is a primary driver of the soil functions that characterize healthy soils, while MAOM is influenced by root exudates and turnover in microbial necromass (Creamer et al., 2019; Lavallee et al., 2020; Hoffland et al., 2020). Therefore, POM is generally more sensitive to changes in land management practices than MAOM (DeGryze et al., 2004; Tan et al., 2007; Eze et al., 2023). For example, Viaud and Kunnemann (2021) associated the decrease in POM with the rapid decline in SOC stocks after hedge removal. Little is known about the changes in SOC fractions with land-use change from grassland to hedges, thus, the distribution of SOC among l-POM, h-POM, and MAOM may provide insights into how hedge planting may influence the quality and persistence of SOC stocks in soils and thus contribution to climate change mitigation.

Studies have reported higher SOC stocks under or near hedges compared to adjacent grassland fields (Ford et al., 2019; Drexler et al., 2021; Viaud and Kunnemann, 2021) and our recent study showed that SOC stocks increase with hedge age until it reaches a plateau around 40 years after planting (Biffi et al., 2022). Using the SOC sequestration rate for 40-year-old hedges, we provided the first estimate of how much CO_2 would be taken up by hedge biomass and stored in soil by increasing total hedge length by 40 % in England (10.53 Tg CO_2 over 40 years, Biffi et al., 2023). To reduce the uncertainty associated with this estimate it is important to consider the influence of pedo-climate on SOC sequestration. To address this knowledge gap, this paper has two main aims: (1) determine the SOC stocks beneath hedges of different ages and in adjacent grassland fields across a range of UK pedoclimatic conditions, and (2) estimate the SOC sequestration rates beneath these hedges. Thus, we quantified SOC stocks within the soil profile at 10 cm intervals up to 50 cm depth under hedges of known age class within five locations representing a range of pedoclimatic conditions across England. We hypothesized that (*i*) SOC stocks would vary across location reflecting the range of climatic and soil type conditions, (*ii*) the amount of SOC stock would be higher beneath hedges than in adjacent grassland fields, (*iii*) the additional SOC stock would depend on the pedoclimatic conditions in which the hedges were found. Finally, we assessed the change in SOC distribution among three soil organic matter (SOM) particle-size fractions to investigate how hedge planting may influence SOC dynamics by affecting the quality and long-term stability of SOC in soils.

2. Methods

2.1. Study sites

Nine farms among five groups were selected to represent a range of pedoclimatic conditions across England, which here will be referred to as five 'locations' (Fig. 1A). The locations were given a name based on their geographic area and based on soil parent material in the form of "*geographic area(parent material)*". England's Koppen climate classification is temperate oceanic, which covers most of Western Europe (Beck et al., 2018). Across England, farms were located within West Sussex (S), Yorkshire (Y), and Cumbria (C). The locations present a range of temperate oceanic climatic conditions: C has mild winters and cooler summers, with high precipitation throughout the year, S has mild winters, warmer summers and lower precipitation than C, and Y has colder winters, mild summers, and less precipitation than C (Met Office, 2022b, Table 1). Each of the five locations presents a different soil parent material: alluvium deposit (AL), chalk (CH), sandstone (GS), mudstone (CL), and dolomitic limestone (LI). Geology, soil type, agricultural land quality (classified based on pedo-climate and topography Natural England, 2010), mean rainfall, temperature, and elevation of each farm are shown in Table 1. The mean slope of fields was 1.7° (range 0–6.5°). The sampled fields were in permanent pasture (81.1 %) or grass leys with the occasional arable crop (18.9 %). Fields were intensively managed, with application of organic and inorganic fertiliser and were cut for silage at least once a year. Grassland was dominated by Italian rye grass (*Lolium perenne*, L.)

2.2. Hedges

Across the nine farms, 52 hedges were selected and grouped into four age categories: (1) '2–5 year old', (2) '10 years old', and (3) '20–30 years old' depending on their known age range, and (4) 'Old' for mature hedges >30 years for which the exact year of planting was not known. The latter category comprised hedges that were planted decades ago to potentially hundreds of years ago. Overall, species composition was typical of North-Western Europe (Barr and Gillespie, 2000) with predominance of hawthorn (*Crataegus monogyna*, 52 %) and blackthorn (*Prunus spinosa*, 32 %), and presence of other species, such as alder (*Alnus glutinosa*, 4 %) hazel (*Corylus avellana*, 4 %), elder (*Sambucus nigra*, 3 %), and wild cherry (*Prunus avium*, 2 %). Most hedges (76 %)

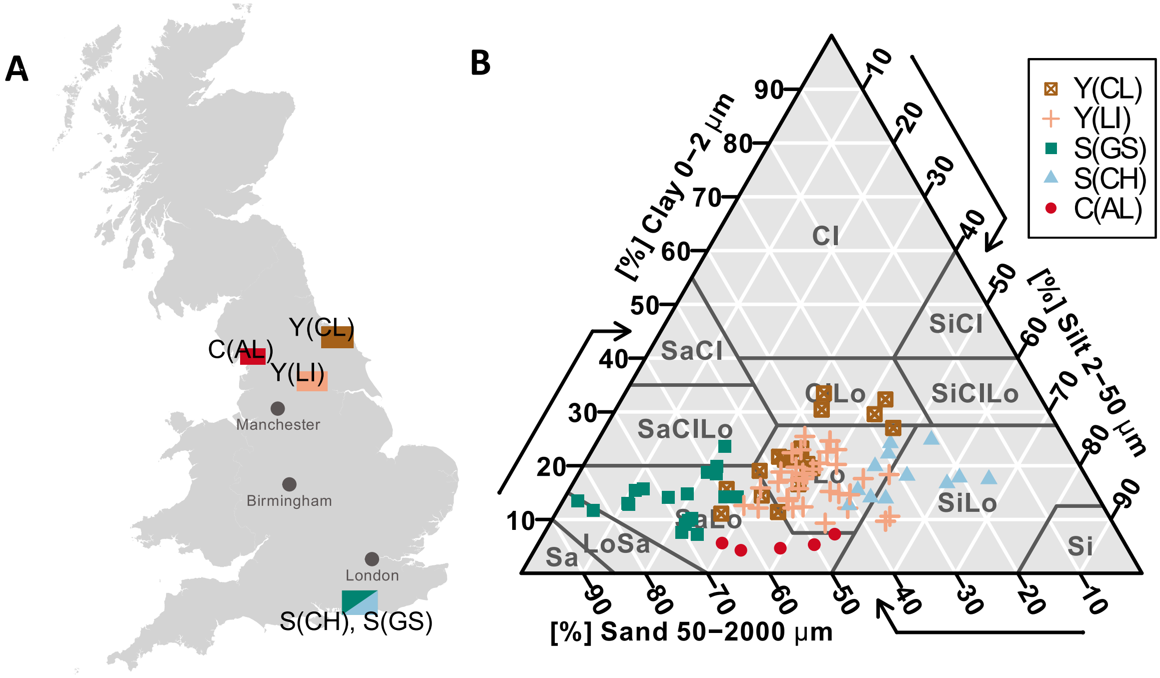

Fig. 1. A–The five locations sampled in this study, with Cumbria (C) characterized by relatively colder and wetter, Sussex (S) by warmer and drier, and Yorkshire (Y) by colder and drier climatic conditions. Alluvium deposit (AL), chalk (CH), sandstone (GS), mudstone (CL), and dolomitic limestone (LI). B–Distribution of soil textures among the five locations of the study in the USDA soil texture triangle.

Table 1
Farms sampled in each location, with their mean annual air temperature and rainfall (Met Office, 2022a), mean altitude, geology and parent material, dominant soil type (camb = cambisol, stag = stagnosol, FAO classification), agricultural land class (ALC, 2 = ‘very good’, 3 = ‘good to moderate’, 4 = ‘poor’, Natural England, 2010), and number of hedges sampled from each of the four hedge age classes, as well as the number of hedge-field pairings (Frac. pairs) that underwent SOM fractionation.

Location	Geology and parent material	ALC	Soil type	Farm ID	Temp (°C)	Rain (mm)	Altitude (m)	3–5 yr	10 yr	20–30 yr	Old	Frac. pairs (n)
C(AL)	Park limestone formation, Alluvium	3	camb	1	9.6	1287	32				3	3
S(CH)	Seafor chalk formation, Chalk	3	camb	2	10.6	930	77		1	2	1	1
				3	10.7	915	44		1		1	
S(GS)	Lower greensand group, Sandstone	3	camb	4	10.7	957	41				4	3
				5	10.6	990	61	4			4	
Y(CL)	Cleveland ironstone formation, Mudstone	4	stag	6	8.9	890	122	2		1	1	2
			camb	7	8.8	938	168			2	1	
Y(LI)	Cadeby formation, Dolomitic limestone	2	camb	8	9.7	688	81	3			3	3
				9	9.8	699	50	5			7	

were fenced with wire mesh for stock-proofing, as prescribed in hedgerow planting agri-environment schemes (RPA, 2022). Hedges ≥ 10 year old were on average 250 cm tall and 195 cm wide, while younger ones were 155 cm tall and 105 cm wide. Mature hedges were managed by regular trimming to maintain a regular height and width and some mature hedges were managed by laying every 15–30 years to increase the density of the shrubs and favour plant rejuvenation (Staley et al., 2015).

Hedges and their adjacent fields were compared using a space-for-time substitution approach (see Biffi et al., 2022). Soil samples were collected between January and March 2022, except for Old hedges and their paired fields at farm 9, which were collected in 2018 (details in Holden et al., 2019). Sampling points under hedges were chosen avoiding gaps, gateways, and tracks, while field sampling points were taken ~16 m perpendicularly from the hedge. Hedges sharing an adjacent field also shared field sampling points. Out of 52 sampled hedges, six were not suitable for the space-for-time substitution approach and were thus removed from further analysis. Of these, two were excluded as the pairings were not located on the same topography (i.e. uphill hedge and downhill field sampling point), resulting in a differential impact of soil drainage on SOC stocks (Walter et al., 2003). The remaining four (Farm ID 6), planted on banks (i.e. mounds elevated about 50 cm from field level), were excluded as the paired fields showed much higher moisture levels and increasing SOC stocks with depth, suggesting a difference in soil type and drainage to that under the hedges. Thus, 46 hedge samples and 35 field samples were included in the analysis, resulting in a total of 46 pairs of samples. A subsample of 12 pairs of mature hedges (20–30 year and Old) and their adjacent grassland fields underwent organic matter fractionation. Table 1 shows the final number of hedges included on each farm.

2.3. Soil sampling and analysis

At each location, a 5 cm diameter ring corer (Eijkelkamp, Holland) was used to take intact 100 cm^3 soil cores at soil depths representative of the layers 0–10, 10–20, 20–30, 30–40, and 40–50 cm for the determination of bulk density and moisture content of each sample. At each depth interval, two grab soil samples were also collected to determine soil organic matter (SOM), C and N content, and soil pH. Moreover, soil samples for analysis of particle size distribution, to determine soil texture, were obtained for the 0–10 cm layer only. We selected pairs of mature hedges and adjacent grassland fields for SOM fractionation of samples at 0–10 and 20–30 cm depth. Maximum soil depth was lower in chalk S(CH) and alluvium C(AL) locations as bedrock was reached at 30 cm, thus the soil profile sampled was 0–30 cm.

2.3.1. Soil physical and chemical characterisation

On return to the laboratory the ring samples were oven dried at 105˚C for 48 h and sieved to <2 mm diameter for the determination of soil moisture content (g g^{-1}) and bulk density (BD, g cm^{-3}). Gravel and roots >2 mm were removed, and their masses recorded. Bulk density was calculated as the difference between the total sample mass and the mass of gravel and roots, divided by the sample volume, following fine soil stock calculations in Poeplau et al. (2017). At each location and sampling depth interval, one of the two grab samples was homogenized, sieved <2 mm), and oven dried at 105˚C before being placed in a furnace for 12 h to determine SOM content (g cm^{-2}) via loss on ignition. The furnace temperature was regulated depending on location: Y(LI) and Y (CL), rich in clay, were processed at 350˚C instead of 550˚C to reduce the loss of structural water (Rowell, 2014). The second grab soil sample was air-dried at 40˚C <2 mm), and then milled to a fine powder using a ball mill (Fritsch Pulverisette agate, Fritsch, Germany) to determine SOC and N content (SOC_{con} and N_{con}, g kg^{-1}). Inorganic C was removed from soil samples by reaction with acid. Samples of <100μm milled soil were placed into 9 × 5 mm silver capsules and 30 μL of 15 % HCl was slowly added. Double strength acid was used for soil from Y(LI) and Y(CL), due to the high bicarbonate present in the soil. The samples were left to react and settle for 24 h and oven dried for 2 h at 80˚C before being analysed for SOC_{con} and N_{con} using an Elemental Vario EL cube (Elementar Analysensysteme GmbH, Germany). For detailed description of the determination of soil pH measurements see Biffi et al. (2022).

2.3.2. Soil organic carbon stocks and sequestration rates

SOC stocks (Mg C ha^{-1}) were estimated using two methods, namely the traditional fixed depth (FD) method to obtain SOC_{FD} stock and the equivalent soil mass (ESM) method to obtain SOC_{ESM}.For detailed description of the SOC_{FD} calculation and of the cumulative coordinate approached used for the ESM correction, see Biffi et al. (2022). Briefly, the FD method calculates SOC_{FD} stocks by multiplying SOC_{con} by bulk density to a fixed depth, while ESM corrections (Wendt and Hauser, 2013; von Haden et al., 2020; Fowler et al., 2023) account for changes in bulk density and SOM following land-use change to calculate SOC_{ESM}. We used model fitting to adjust SOC_{ESM} stocks under hedge to the reference cumulative mineral mass in the fields, using a cumulative coordinate approach (Gifford and Roderick, 2003; Wuest, 2009) and assumed exponentially decaying SOC_{con} through the soil profile (Rovira et al., 2015; Murphy et al., 2019, Figure S1.1). SOC_{FD} results are shown in Table S4.1 and are specified where necessary for comparison with previous studies, but from now onwards we will refer exclusively to SOC_{ESM} stock as SOC stock.

We calculated the additional SOC stock accumulated over time as a result of planting hedges (ΔSOC) as the difference between SOC stocks in the hedges and fields. Because the sample size by age within each location was small, we calculated the ΔSOC by location across all hedge age categories. ΔSOC estimates per age group and across all locations are presented in the discussion.

Mean annual SOC sequestration rate was calculated by dividing the ΔSOC of each individual hedge by the number of years since planting, which was assumed to be 4, 10, 25, and 50, respectively, for 2–5 years, 10 years, 20–30 years, and Old hedges. SOC stocks and annual sequestration rates were reported in Mg C ha^{-1}, as well as by length of hedge (Mg C km^{-1}) assuming a hedge width of 1.5 m (i.e., by dividing estimates per hectare by 6.67), the prescribed planting width within AES (RPA, 2022). Moreover, CO_2 equivalent results were calculated by multiplying Mg C by the ratio of molecular weight of CO_2 to C (ratio = 3.67).

2.3.3. Particle size analysis

Soil particle size was determined by adapting the gravimetric method described in Van Reeuwijk (2002). Briefly, 100 ml of deionized water was added to 10 g of air-dried and homogenized soil. A 5 % sodium hexametaphosphate solution (Calgon) was added and left overnight to disaggregate the soil mineral fine fractions. The suspension was passed through a 53 μm sieve. The percentage of sand fraction (53–2000 μm) was obtained by wet sieving, while the percentage of clay fraction <2μm) was obtained with the pipette method. Silt content was determined by subtraction.

2.3.4. Organic matter fractionation

To investigate differences in soil C pools under mature hedges compared to fields, SOM was fractionated following Robertson et al. (2019). Briefly, 5 g of <2 mm oven-dry soil was density fractioned by adding 20 ml of sodium polytungstate (SPT, 1.85 g cm^{-3}) in a 25 ml centrifuge tube, shaking (18 h at 95 rpm), and centrifugating (30 min at 2500 rpm). After centrifugation, the floating l-POM <1.85 g cm^{-3}) was suctioned off using a vacuum flask and collected on a 20 μm paper filter. Deionized water was added to the soil sample remaining in the centrifuge tube, which was then shaken and centrifugated before discarding the water. This process was repeated twice to rinse SPT and remove dissolved organic matter. Heavy POM >53μm–2000 μm) and MAOM <53μm) were separated by wet sieving. The mass of each fraction was recorded as a proportion of total soil mass ($mass_{frac}$, %). The recovery rate across all samples was 98.9 ± 1.0 %. Fractions were then oven-dried at 60˚C and analysed for SOC_{con} and N_{con} on an elemental analyser as described above for bulk soils. The SOC_{con} of the l-POM, h-POM, and MAOM fractions will be referred to as l-POC, h-POC, and MAOC, respectively. The element content (E_{con}) of the three respective fractions were adjusted for total soil mass using Equation (1).

$$E_{con}(\text{g kg}^{-1}\text{ of soil}) = \frac{E_{con}(\%) \times \text{mass}_{\text{frac}}(\%)}{10} \tag{1}$$

The effect size of the land-use change from grassland to mature hedges on the SOC_{con} and N_{con} of the three SOM fractions was estimated following the method by Eze et al. (2023) and adapted from Hedges et al. (1999), using Eq. (2).

$$\text{Effect size} = \ln\frac{\text{Hedge mean } E_{con}}{\text{Field mean } E_{con}} \tag{2}$$

The effect size was also expressed as a percentage using Eq. (3).

$$\text{Effect size}(\%) = 100 \times \exp(\text{Effect size} - 1). \tag{3}$$

2.4. Data analysis

All analyses were conducted in R v4.2.3 (R CoreTeam, 2023). Differences in pH, bulk density, SOC_{con}, N_{con}, SOC stock, and SOM fractions among locations and beneath hedges and in adjacent fields were investigated using Student t-tests and ANOVA or, when the data distribution did not meet the assumption of normality, non-parametric Mann-Whitney-Wilcoxon tests and Kruskal-Wallis rank tests. Post-hoc tests were conducted with Tukey's Tests or non-parametric Dunn's tests for pairwise multiple comparisons with Benjamini-Hochberg false discovery rate-corrected p-values. Differences were considered significant at $\alpha = 0.05$.

Associations between SOC stock and SOC sequestration rates with hedge age and other explanatory variables were investigated using a linear model. SOC stocks under hedges and in fields were also modelled separately. Continuous predictors of sample depth, silt and clay fractions, soil N_{con}, soil pH, altitude, rainfall, and temperature were considered as covariates. In the SOC sequestration rate model, soil N stock (calculated using a cumulative coordinate approach as described in Section 2.3.2) was used instead of N_{con} while sample depth was excluded from the predictors, as sequestration was calculated for the entire soil depth. Prior to modelling predictors were tested for multicollinearity and pH and altitude were excluded from the analysis due to high Pearson correlation coefficients with rainfall and temperature, respectively (Figure S2.1). Predictors were normalised for comparability of effect size. SOC stocks were square-root transformed and sequestration rates were log-transformed, as the data had unequal variance. Model assumptions were checked with residual plotting and assumptions were met.

Unless otherwise specified, ranges in brackets indicate 95 % confidence intervals around the mean.

3. Results

3.1. Differences in soil characteristics between hedges and fields

Soil beneath hedges had significantly lower bulk density (1.15 *vs* 1.28 g cm^{-3}) and higher SOC_{con} (29.41 *vs* 20.71 g kg^{-1}) compared to soil in adjacent fields (Table S3.1). SOC_{con} was consistently higher under hedges than in fields for all locations except for C(AL), where it showed no significant difference. Across all locations, soil beneath hedges had significantly higher N_{con} and higher C:N ratio than soil in adjacent fields. Significant differences in soil bulk density between hedges and fields were found in locations S(GS) and Y(LI), while S(CH) was the only location showing a difference in soil moisture content, which was higher beneath hedges than in adjacent fields. Soil pH did not significantly differ between hedges and fields.

The samples ranged in soil texture, with a wide range in sand (15.9 %–84.1 %) and silt (2.4 %–66.6 %) content and a narrower range in clay content (4.3–33.5 %, Fig. 1B, Table S3.2). Clay and silt fractions did not differ significantly between hedges and fields in the same location, confirming that hedges and fields were paired on the same soil types.

3.2. Soil organic carbon stocks in fields and under hedges

Mean SOC stocks of grassland fields in different locations ranged from 74.4 Mg C ha^{-1} in S(GS) to 125.9 Mg C ha^{-1} in Y(CL) (Fig. 2A). Mean field SOC stock in S(GS) was significantly lower than in Y(LI) and Y(CL). All other location comparisons were not significantly different.

SOC stocks beneath hedges across all locations and ages were significantly higher than in adjacent fields (Fig. 2A, tabular results in Table S4.1). Mean SOC stock (0–50 cm) under hedges of all ages ranged between 108.1 Mg C ha^{-1} in S(GS) and 216.6 Mg C ha^{-1} in Y(CL). On average, soil under hedges of 10 years and younger stored significantly more SOC than in fields down to 20 cm depth, while this difference was significant down to 40 cm depth under hedges older than 20 years (Fig. 2B).

There was no significant difference in additional SOC stock (ΔSOC)

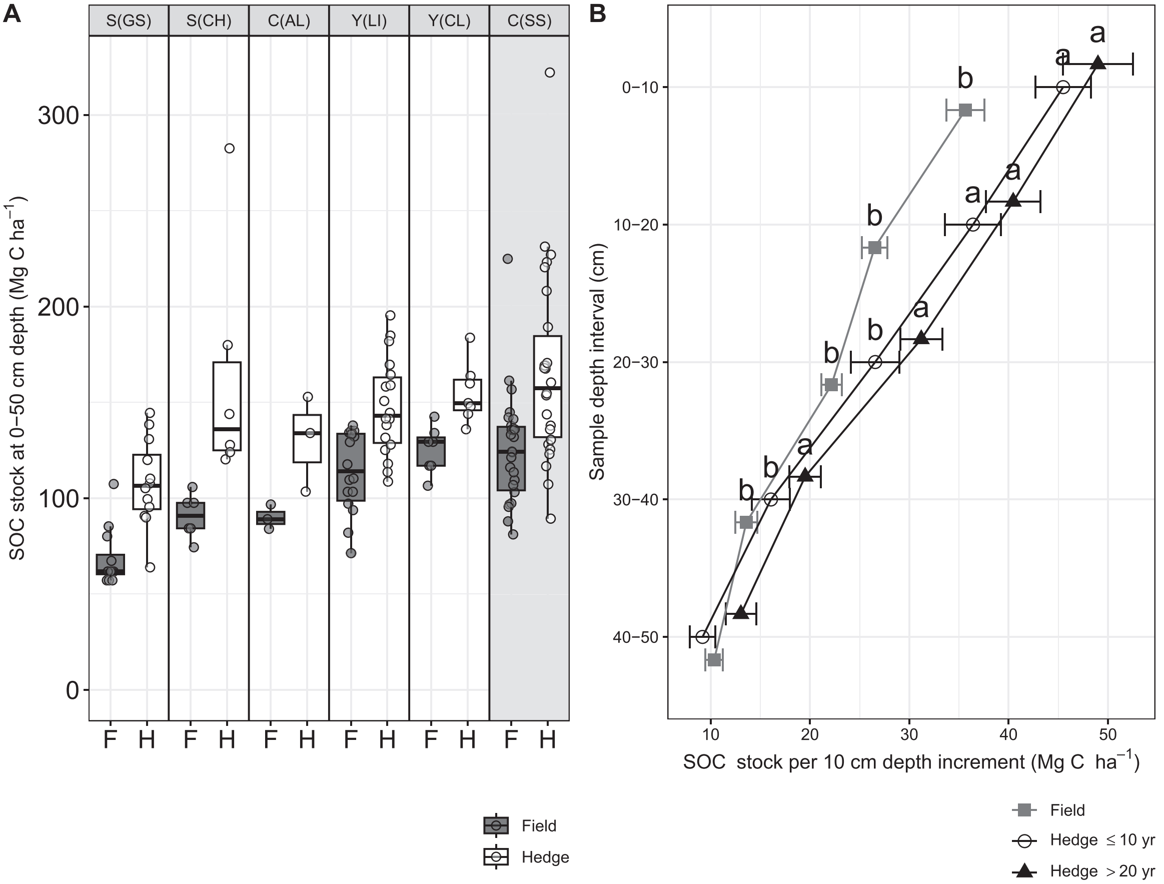

Fig. 2. Comparisons of soil organic carbon (SOC) stocks at 0–50 cm depth or to bedrock under hedges and in adjacent grassland fields according to equivalent soil mass correction. A– Boxplots showing the median and the 25*th* and 75*th* percentiles of SOC stocks under hedges (H) and in fields (F). The first five panels show the five locations sampled in this study. The shaded panel shows the distribution of SOC stocks in our previous study (Biffi et al., 2022) conducted in Cumbria (C) on sandstone parent material (SS). The C(SS) data is shown to facilitate visual comparison with the locations included in this study and is not considered in the analysis. B– Mean (± SE) SOC stock down the soil profiles under recently planted hedges (≤10 years), under mature hedges (> 20 years since planting), and in adjacent predominantly grassland fields. Each point represents the SOC stock measured within a 10 cm depth interval, with vertical jitter to facilitate readability. Different letters indicate statistically significant differences according to post-hoc Dunn's tests comparing hedges and fields within individual soil depth intervals ($p < 0.05$).

among locations. Mean ΔSOC was 40.5 (30.6–50.4) Mg C ha^{-1} in the top 50 cm of soil, or 6.1 Mg C km^{-1} assuming a hedge width of 1.5 m. Hedges stored an additional 39.5 % SOC stock compared to fields at 0–50 cm depth and 29.7 % at 0–30 cm. 77 % (34.5 Mg C ha^{-1}, or 5.2 Mg C km^{-1}) of the additional SOC stock was found in the top 30 cm of the soil profile. No significant difference in ΔSOC among locations was found both when considering the ΔSOC of all hedge ages, nor when considering ΔSOC of only hedges $\geq$ 20 year old (result not shown).

The linear models of SOC stocks of individual sampling depth intervals showed that field SOC stocks were positively associated with both silt and clay content, while hedge SOC stocks were only associated with silt content (Fig. 3A). Sample depth was negatively associated with SOC stock, while soil N$_{con}$ was positively associated with SOC stocks. Field SOC stocks were also higher under drier and colder conditions. Soils under hedges were associated with higher SOC stock than adjacent fields for all age categories (Fig. 3B, Tables S5.1–3).

3.3. Soil organic carbon sequestration rates under hedges

The mean annual SOC sequestration rate declined rapidly with age, with younger hedges showing higher variability. Sequestration rates decreased and became less variable as hedges matured. Table 2 presents sequestration rates per hedge age across all locations and by location. The linear regression model showed that SOC sequestration declined with increasing hedge age, and that higher SOC sequestration rates were associated with higher N stocks. Climatic conditions and soil texture, instead, did not influence sequestration rates (Fig. 3C).

3.4. Organic C and N distribution by SOM fractions in soil under mature hedges

Across all samples, l-POM on average accounted for 1.88 (1.59–2.17) % of bulk soil but was on average 55 % higher under hedges (2.37, 1.81–2.92 %) than in adjacent fields (1.52, 1.25–1.80 %, KW $\chi^2 = 8.47$, $p = 0.004$). This difference was significant at both sampling depths, with l-POM 134 % higher under hedges than in adjacent fields at 0–10 cm depth (mean field = 1.74, 1.32–2.15; hedge = 3.09, 2.23–3.95; KW $\chi^2 = 13.4$, $p < 0.001$) and 17 % higher at 20–30 cm depth (mean field = 1.31, 0.95–1.68; hedge = 1.64, 1.11–2.18; KW $\chi^2 = 9.95$, $p = 0.002$). The h-POM and MAOM fractions accounted on average for 40.83 (40.13–40.54) % and 56.20 (55.92–56.51) % of bulk soil, respectively, and did not change significantly between hedge and field samples.

l-POM had the highest SOC$_{con}$ (l-POC = 165.5, 139.0–191.9 g kg^{-1} of l-POM). However, as it accounted for the smallest fraction in the bulk soil, once adjusted for bulk soil mass, l-POC was 3.17 (2.20–4.14) g kg^{-1}, h-POC was 8.35 (6.27–10.40), and MAOC was 24.90 (22.01–27.79) g kg^{-1}. l-POC, h-POC, and MAOC did not change significantly among locations, although MAOC was lower in S(GS) than other locations (Figure S6.1). Together, l-POC and h-POC contributed to 33 % of SOC under hedges and 23 % in grassland fields, with l-POC representing 12 % and 4 % of the total SOC pool under hedges and in grassland, respectively. The l-POC under hedges (4.73, 2.99–6.46 g kg^{-1}) was significantly higher (+194 % increase) than in fields (1.61, 1.13–2.09 g kg^{-1}, KW $\chi^2 = 15.5$, $p < 0.001$), increasing by a mean of 3.12 (1.39–4.84) g kg^{-1}, or 4.47 (1.25–7.69) g kg^{-1} at 0–10 cm and 1.68 (0.25–3.12) g kg^{-1} at 20–30 cm depth. h-POC and MAOC did not differ

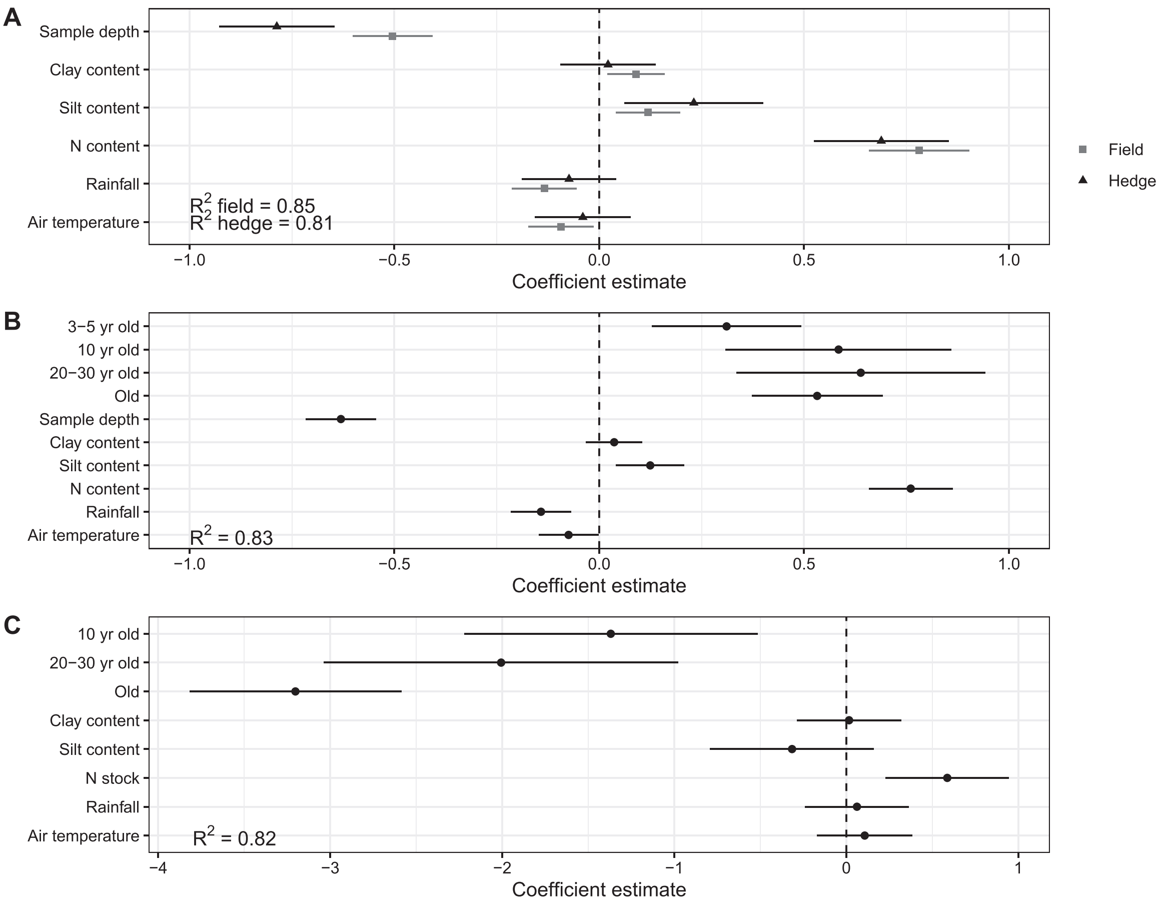

Fig. 3. Dot and whisker plots showing significance of the predictors of SOC stocks in fields and under hedges and SOC sequestration rates under hedges using linear regression. Scaled model coefficients are shown with 95 % confidence interval whiskers for A– field and hedge SOC stocks modelled separately, B– SOC stocks estimate by field and hedges of different age categories, and C– SOC sequestration rate under hedges of different ages. Effect sizes are significant only when confidence intervals do not overlap with the dotted line indicating zero.

Table 2

Estimated mean annual SOC sequestration rates (and 95 % confidence intervals where n > 2), calculated on an equivalent soil mass basis, beneath hedges across all locations and within each location. Values refer to 0–50 cm depth or to bedrock depth (0–30 cm for S(CH) and C(AL)). Values expressed in km refer to a hedge of 1.5 m width, the prescribed planting width for new hedges (RPA, 2022).

Location	Hedge age	n	Mg C ha^{-1} yr^{-1}	Mg C km^{-1} yr^{-1}	Mg CO_2 km^{-1} yr^{-1}
All	2–5 yr	14	12.42 (8.71–16.12)	1.86 (1.31–2.42)	6.84 (4.79–8.88)
	10 yr	6	5.22 (2.01–8.43)	0.78 (0.30–1.26)	2.87 (1.11–4.64)
	20–30 yr	5	1.49 (0.93–2.05)	0.22 (0.14–0.31)	0.82 (0.51–1.13)
	Old	21	0.80 (0.40–1.21)	0.12 (0.06–0.18)	0.44 (0.22–0.66)
C(AL)	Old	3	0.80 (-0.68–2.28)	0.12 (-0.10–0.34)	0.44 (-0.37–1.26)
S(CH)	10 yr	2	5.03	0.75	2.77
	20–30 yr	2	1.59	0.24	0.88
	Old	2	2.54	0.38	1.40
S(GS)	2–5 yr	4	12.25 (0.37–24.13)	1.84 (0.06–3.62)	6.74 (0.20–13.28)
	10 yr	4	5.32 (0.63–10.01)	0.80 (0.09–1.5)	2.93 (0.35–5.51)
	Old	4	0.58 (0.23–0.93)	0.09 (0.04–0.14)	0.32 (0.13–0.51)
Y(CL)	2–5 yr	2	11.49	1.72	6.33
	20–30 yr	3	1.42 (-0.08–2.92)	0.21 (-0.01–0.44)	0.78 (-0.05–1.61)
	Old	2	0.34	0.05	0.18
Y(LI)	2–5 yr	8	12.73 (6.82–18.64)	1.91 (1.02–2.80)	7.01 (3.75–10.26)
	Old	10	0.64 (0.26–1.02)	0.1 (0.04–0.15)	0.35 (0.14–0.56)

significantly between hedge and grassland (Fig. 4A). The effect size of l-POC difference from grassland to mature hedges was 288 % (Fig. 4B).

Similarly, N content of l-POM was 8.58 (6.97–10.20) g kg^{-1} of l-POM and 0.17 (0.12–0.23) g kg^{-1} of bulk soil. The l-POM N_{con} of soil under hedges (l-PON, 0.26, 0.17–0.35 g kg^{-1}) was significantly higher than in adjacent fields (0.08, 0.05–0.12 g kg^{-1}, KW $\chi^2 = 14.63$, $p < 0.001$), increasing by a mean of 0.17 (0.07–0.27) g kg^{-1}. l-PON and MAON did not change between hedge and field samples. The effect size of l-PON difference from grassland to mature hedges was 320 %.

SOC_{con} and N_{con} were significantly higher at 0–10 cm depth than at 20–30 cm in all three fractions ($p < 0.001$). Similar trends in across the three fractions were found at both 0–10 and 20–30 cm depth and in each of the five locations sampled in this study. Results by depth and by location are shown in Figure S6.1–3.

The three fractions differed in C:N ratio, with the ratio of l-POM (20.33, 18.95–21.70) being higher than that of h-POM (13.00, 11.70–14.31), and h-POM higher than MAOM (10.31, 9.96–10.66), with the same trend observed for both soil depths and beneath hedges and in fields. The C:N ratio beneath hedges and in grassland did not differ in l-POM and h-POM, while in the MAOM fraction it was higher under hedges (10.88, 10.40–11.38) than in fields (9.73, 9.32–10.13, KW $\chi^2 = 13.80$, $p < 0.001$).

4. Discussion

4.1. *SOC stocks beneath hedges and in fields across pedoclimatic conditions*

Our results showed that intensively managed grassland field SOC stocks generally did not significantly change across locations, and mean field SOC stock across locations was 103.0 Mg C ha^{-1} at 0–50 cm depth or to bedrock. Field SOC stocks ranged from 74.4 (57.8–90.9) Mg C ha^{-1} in the coarse textured greensand C(GS) to 125.9 (108.2–143.6) Mg C ha^{-1} in Y(CL). SOC stocks have been quantified at a range of depth intervals across UK grassland; as we have quantified SOC stocks at 10 cm depth intervals our results can be compared to a range of other empirical studies. The field mean SOC stock in our study was generally consistent with that previously reported for UK grasslands. For example, Bradley et al. (2005) reported SOC stocks of 80 Mg C ha^{-1} at 0–30 cm depth in pasture, which is very similar to the 84.32 Mg C ha^{-1} we found for the same depth interval across all locations. Similarly, Beka et al. (2022) reported a measured SOC stock of 38.2 Mg C ha^{-1} at 0–10 cm depth (when excluding predominantly silvopasture sites), compared to our 35.6 (11.2–31.7) Mg C ha^{-1}. Ward et al. (2016) reported 82.6 Mg ha^{-1} for intensive grassland at 0–20 cm depth, higher than our mean of 62.2 (56.5–67.8) Mg ha^{-1} at the same interval. Moreover, the range of stocks across locations was representative of the range previously shown in UK grassland, with the upper limit very similar to the 124.9 Mg C ha^{-1} reported for cambisol and stagnosol grassland (Biffi et al., 2022) and the lower limit within the range (58.9–100.7 Mg C ha^{-1}) reported for sandy loam (arenosol) grassland (Eze et al., 2018b). While sandy soils are expected to have low SOC stocks (Cordeiro et al., 2022), the lack of a significant difference among the other locations can likely be explained with a combination of limited range in climatic variation and homogenisation of land-use management. The locations included in our study are representative of the range of climatic conditions across which improved grasslands are found in England (Ward et al., 2016). Climatic conditions are known drivers of grassland SOC stocks (Hewins et al., 2018; Poeplau, 2021); however, the climate of the whole of the UK is classified as ‘temperate oceanic’ (Beck et al., 2018). Thus, climate signals across locations were relatively narrow. SOC stocks in UK grasslands are more strongly affected by management practices than climate (Ward et al., 2016) and fields in this study were characterised by the homogeneous intensive agricultural grassland management.

The main driver of SOC stocks across the locations was land-use, as shown by the consistently higher SOC stock observed under hedges compared to adjacent improved grassland fields. The mean SOC stock at 0–50 cm depth beneath hedges was 139.2 Mg C ha^{-1} (132.0 Mg C ha^{-1} for hedges ≤10 years old and 144.8 for mature hedges ≥20 year old). The estimated UK forest SOC_{FD} stock of cambisols and gleysols ranges between 135–155 Mg C ha^{-1} at 0–80 cm depth (Vanguelova et al., 2013); thus, our results indicate that soil under mature hedges can store SOC stocks comparable to forested areas. The linear model of SOC stock in fields and under hedges of different ages showed that hedges of all ages stored higher SOC stocks than adjacent grassland fields. Model estimates for 10 year and 20–30 year old hedges are higher than Old hedges. However, this result should be considered carefully as the sample size of 2–5 year old (n = 14) and Old (n = 21) hedges was greater than the other two age categories (n = 6 and 5, respectively), resulting in higher error estimates in 10 and 20–30 year old hedges, as indicated in Fig. 3B. The model estimate of SOC stocks under Old hedges was higher than that of 2–5 year old ones, indicating a progressive accumulation of SOC stocks over time as the hedge grows and matures, a result in line with Biffi et al. (2022).

When accounting for climate and soil covariates in the models of SOC stocks, differences in temperature, rainfall and soil texture explained some of the variability in total SOC stocks observed in individual grassland fields and under hedges. Soil texture had an effect on SOC stocks under hedges and in fields, while climatic conditions were associated with lower SOC stocks in grassland fields, but not under hedges. This suggests that, under woody vegetation, soil texture was a stronger driver of SOC stock than climate. Soil texture, particularly silt content, played a significant role in controlling SOC storage. In fields, silt content had a larger effect size than clay, likely due to the locations included in our study having a greater range in silt content than clay content. All soils were rich in clay (> 33 %), suggesting a good capacity to accumulate SOC (Laganière et al., 2010). Thus, a larger amount of model variance was explained by silt content. Moreover, silt and clay can

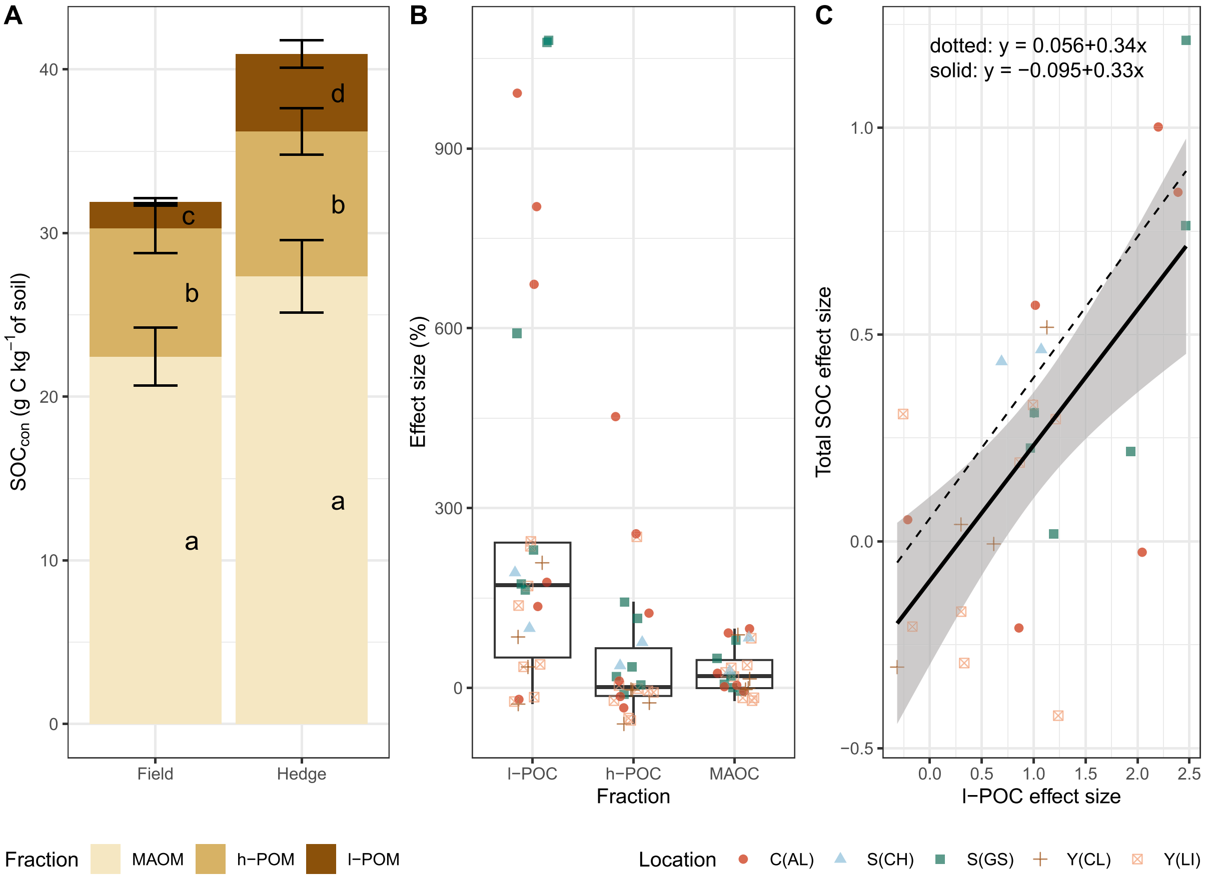

Fig. 4. Results of the SOM fractionation analysis. A– Bar chart showing the mean (±SE) SOC$_{con}$ across three particle size fractions (l-POM, h-POM, MAOM) in soil under mature hedges (n = 13) and in adjacent grassland fields (n = 13), averaged between measurements at 0–10 and 20–30 cm depth. Different letters indicate statistically significant differences according to Dunn's post-hoc tests ($p < 0.05$). B– Boxplot showing the median and 25th and 75th percentiles of the effect size of C$_{con}$ change with the land-use change from grassland to hedge for each of the three SOM fractions calculated as the natural logarithm of the ratio of hedge and field mean SOC$_{con}$, following Eze et al. (2023) (Eq. (2) and (3)). C– Scatterplot showing the relationship between effect size of change in total SOC with the land-use change from grassland to hedge with the effect size of l-POC change. The solid line with the grey band represents the regression line and 95 % confidence intervals in the five locations of this study and the dotted line shows the regression line in the global meta-analysis by Eze et al. (2023).

present different types of associations with SOC, as, for examples, lignin-derived SOC has been shown to preferentially accumulate and preserve in the silt fraction of the soil (Heim and Schmidt, 2007). The regular trimming management of hedges leaves woody lignin-rich residues to decompose on the ground, which likely contributed to the lack of a significant association with clay fine mineral soil fraction. It should be noted that particle size distribution was determined only for the 0–10 cm layer and that it might change down the soil profile, as found by Antony et al. (2022).

SOC stocks of individual soil depth intervals decreased with depth, and this negative association was stronger under hedges than in fields (Figs. 2B, 3A). While Wenzel et al. (2023) found no difference in SOC stocks under 30–70 year old hedges below 20 cm depth when compared to arable fields, we found that soil under younger hedges (≤10 years) was higher in SOC than grass fields up to 20 cm depth and that under mature hedges (≥20 years) this difference was significant to 40 cm depth. Our results align with the findings of previous studies that indicate that hedges positively influence SOC stocks to depth (e.g. Cardinael et al., 2017; Viaud and Kunnemann, 2021; Chiartas et al., 2022; Mayer et al., 2022; Lesaint et al., 2023).

4.2. *Additional SOC stocks beneath hedges across pedoclimatic conditions*

Vegetation type, rather than pedoclimate, had the strongest effect on SOC stocks in the study, and the additional SOC stocks beneath hedges compared to adjacent grassland (ΔSOC) were not significantly different among the five locations. The ΔSOC in our study was 40.5 Mg C ha^{-1} at 0–50 cm depth and 34.5 Mg C ha^{-1} at 0–30 cm. Few studies have quantified SOC stocks beneath woody linear features and adjacent agricultural fields, and fewer still have provided comparisons between different soil types and/or climatic conditions. The mean ΔSOC found in our study were very similar to those reported by previous studies comparing SOC stocks between hedges and grasslands. For example, in our previous study we found an additional 41.5 Mg C ha^{-1} stored under hedges on cambisols and stagnosols in Cumbria (0–50 cm, sandstone parent material, Biffi et al., 2022). A visual comparison with the SOC stocks measured in Biffi et al. (2022) has been provided in Fig. 2A. In France, Follain et al. (2007) reported ΔSOC of 33 Mg C ha^{-1} and Viaud and Kunnemann (2021) of 34.4 Mg C ha^{-1} at 0–30 cm depth, 1 m away from hedges compared to further into the adjacent field. A meta-analysis of agroforestry also found an additional SOC stock of 41.5 Mg C ha^{-1} compared to alley cropping at 20–40 cm depth (Mayer et al., 2022). Similarly, (Drexler and Don, 2024) found SOC stocks under hedges to be significantly higher than in cropland fields to 100 cm depth under a range of pedoclimatic conditions across Germany. Importantly, a recent study in the USA observed that hedges planted in warm and dry climatic conditions stored an additional SOC$_{FD}$ stock of 38 Mg C ha^{-1} at 0–100 cm depth compared to adjacent cropland fields (field SOC$_{FD}$ stock = 106 Mg C ha^{-1}), and that this additional SOC$_{FD}$ stock did not vary across four soil types (Chiartas et al., 2022).

Soils beneath hedges stored 40 % and 30 % more SOC stock compared to adjacent grassland fields at 0–50 and 0–30 cm depth, respectively. The percentage increase in SOC stock in our study is in the same range as that reported in recent meta-analyses for the conversion of agricultural soils to agroforestry. For example, De Stefano and Jacobson (2018) found 34–40 % increase in stock at 0–30 and 0–100 cm depth, Guo et al. (2021) 46–52 % at 0–20 and 20–60 cm depth, and Eze et al. (2023) 34–47 % at 0–100 cm depth. It should be noted that the C stock of soil litter was not measured in this study. A study in Germany found hedge litter to contain 2 Mg C ha^{-1} (Drexler et al., 2023), half as much as forestry litter estimates in Western Europe (Jarmain et al., 2023; Drexler et al., 2023).

4.3. Mean and net SOC sequestration beneath hedges

This study gives representative values of SOC sequestration for hedges across England, which can be used for future climate change mitigation modelling and carbon farming calculations. Our results highlight that SOC accumulation following hedge planting is non-linear, as it is more rapid during the early years. After the hedge has been in place for decades the SOC stock tends to reach a new equilibrium, in which the carbon input is equal to the carbon released by the mineralization of organic matter (Fujisaki et al., 2018; Rumpel et al., 2020). For hedges in England this appears to be ~40 Mg C ha^{-1} more than the adjacent grassland.

Mean SOC sequestration rates did not vary significantly between locations but decreased substantially over time. Mean annual SOC sequestration for Old hedges was 0.80 Mg C ha^{-1} yr^{-1} and ranged from 0.34 to 2.54 Mg C ha^{-1} yr^{-1}. Few studies have attempted to estimate SOC sequestration rates under managed hedges, so comparisons are difficult, but this range is comparable to the SOC_{FD} 0.46 Mg C ha^{-1} yr^{-1} (0–30 cm depth) and 2.7 Mg C ha^{-1} yr^{-1} (modelled from field observations of mature, unmanaged hedges) reported by Robertson et al. (2012) and Crossland (2015), respectively. The mean SOC sequestration rate of 20–30 year old hedges was 1.42 (Y(CL)) and 1.59 (S(CH)) Mg C ha^{-1} yr-1, results remarkably close to the 1.48 Mg C ha^{-1} yr^{-1} in 37 year old hedges for hedges in Cumbria (Biffi et al., 2022). Young hedges showed high sequestration rates, likely due to the addition of mulch and mixing of the soil profile when planting. These practices can result in biased sequestration estimates (Laganière et al., 2010) that should be considered carefully. However, hedges in this study were planted in commercial settings and these values are representative of standard planting practices in the country.

While mean annual sequestration rates estimate the sequestration rate of existing hedges over a set number of years, net sequestration rates are estimates of the changing sequestration rate of hedges from planting to maturity, thus they are essential in forecasting the climate changing mitigation potential of hedge planting (Nair, 2012). Table 3 shows estimates of additional SOC stock and the time series of the change in SOC sequestration rate from planting to maturity from this study, together with results from Cumbrian hedges planted on sandstone geology presented in Biffi et al. (2022). The mean additional hedge SOC stock across hedges in the six locations is 40.89 (32.92–48.85) Mg C ha^{-1} across all age categories. We then calculated the mean annual SOC sequestration rate and the net annual sequestration rate of hedges of known ages. Net annual sequestration was obtained by dividing the change in mean SOC stock between subsequent age categories by their average age difference (4, 6, and 22 years). These estimates represent the SOC sequestration potential of hedges from planting to maturity and indicate that SOC sequestration declines from 7.6 to 1.5 Mg C ha^{-1} yr^{-1} (27.9–5.6 Mg CO_2 ha^{-1} yr^{-1}) during the first decade after planting, and then to 0.4 Mg C ha^{-1} yr^{-1} (1.5 Mg CO_2 ha^{-1} yr^{-1}) during the second to fourth decade after planting. Our results show that net SOC sequestration rates after planting new hedges are consistent across the country. These net SOC sequestration estimates can be used in carbon farming calculations, as well as in future scenario modelling to assess the contribution of different hedgerow planting targets towards climate change mitigation and 2050 net-zero goals in England (Climate Change Committee, 2018; Burgess and Graves, 2022; Biffi et al., 2023).

4.4. l-POM as the driver of SOC change under hedges

Our findings indicate that most of the SOC sequestered by soil under hedges is stored in the l-POM fraction of SOM. l-POM, the SOM fraction dominated by free plant material, including litter and root inputs (Llorente et al., 2017; Lal, 2018), was the only fraction that increased significantly with the land-use change from grassland to hedge, particularly in the top 10 cm of soil (134 % increase). This result is in accordance with previous agroforestry and afforestation studies that showed that the change from grassland to permanent woody plant cover led to an increase in POM (Six et al., 2002; Arevalo et al., 2009; Poeplau and Don, 2013; Wei et al., 2013; Wenzel et al., 2023), due to greater annual litter biomass and root turnover beneath woody vegetation compared to grass (Axe et al., 2017; Cardinael et al., 2018; Witzgall et al., 2024). For example, Hernandez-Ramirez et al. (2011) reported that 54 % of the SOC found underneath coniferous shelterbelt hedges was derived from tree biomass. Our results thus indicate that changes in POM, and l-POM in particular, can be used as an indicator of short-term changes in SOC stocks under hedges, a result in accordance with the findings of a global meta-analysis of the effects of agroforestry practices on SOC sequestration (Eze et al., 2023).

Together with the l-POM fraction, its SOC content (l-POC) increased by 194 % under hedges compared to fields. On average, the effect size of l-POC, calculated following Eze et al. (2023), was 288 % and reached a maximum of 1080 %. For comparison, Poeplau and Don (2013) found increases in l-POC up to 2000 % with the land-use change from agriculture to woodland. Interestingly, the regression of total SOC with l-POC effect size in Fig. 4C displayed a marginally different intercept but a very similar slope to that found by Eze et al. (2023) in a global meta-analysis of different agroforestry practices compared to grassland. The difference in intercept may be explained by the inclusion of different agroforestry practices in Eze et al. (2023), including the afforestation of arable soils. However, the similarity in slope suggests a predictable change in l-POC with the land-use change from grassland to woody permanent vegetation in agricultural settings, which results in changes in soil structure and increase in litter and root inputs (Guest et al., 2022; Eze et al., 2023). Finally, we found that l-POC was greater under hedges than in adjacent fields at both sampling depths, a result in accordance with a recent study conducted in England, which showed that the POM fraction remained greater in woodland soil than in grassland soil down

Table 3

Mean (and 95 % confidence intervals) SOC stocks, additional stock (ΔSOC), and sequestration rates (SOC_{seq}) beneath hedges of different age categories as calculated from the results of this study combined with the results presented in Biffi et al. (2022), and the respective net sequestration rate (Net SOC_{seq}) showing the change in SOC stock over time between age categories. Values expressed in km refer to a hedge of 1.5 m width, the prescribed planting width for hedges (RPA, 2022).

Hedge age	n	SOC stock	ΔSOC	SOC_{seq}	Net SOC_{seq}		
		Mg C ha^{-1}	Mg C ha^{-1}	Mg C ha^{-1} yr^{-1}	Mg C $ha^{-1}yr^{-1}$	Mg C $km^{-1}yr^{-1}$	Mg CO_2 $km^{-1}yr^{-1}$
2–5 yr	19	132.6 (117.0–148.3)	30.4 (20.1–40.6)	7.6 (5.0–10.1)	7.6	1.1	4.2
10 yr	12	135.1 (117.0–148.3)	39.6	3.9	1.5	0.2	0.8
20–40 yr	13	168.8 (117.0–148.3)	48.0 (31.4–64.5)	1.5 (1.0–2.0)	0.4	0.1	0.2

to 100 cm depth (Antony et al., 2022).

We found that the SOC content of the fractions appeared to be driven by biomass input rather than pedoclimatic conditions. In general, pedoclimatic conditions did not influence the SOC content of the three fractions, except for a (not significantly) lower MAOC content in S(GS). Our results are in accordance with Eze et al. (2018a), who found no effect of soil parent material or climate on POC and MAOC in UK semi-natural grasslands. Unsurprisingly, we found that MAOM increased with depth and represented the largest of the three SOM fractions (Antony et al., 2022), and that its SOC_{con} (MAOC) was the greatest SOC pool in the soil both under hedges and in fields. MAOC stock has been shown to represent around 60 % of total SOC stock in UK grassland (Eze et al., 2018b) and in our grassland fields MAOC made up 70.5 % of SOC across the three fractions. This estimate is within the typical 50–75 % range reported for grassland (Cotrufo et al., 2019; Bai and Cotrufo, 2022; Guillaume et al., 2022). Under hedges, MAOC content was very similar to fields and represented 67.0 % of the element pool, a result very similar to that found by Viaud and Kunnemann (2021) 1 m away from mature hedges in France (~68 %).

As POM is vulnerable to disturbance and land-use changes (Cotrufo et al., 2019) it is crucial that hedges are well managed and maintained over time. The stability of POM can change depending on land-use, as, for example Cambardella and Elliott (1992) found that the stable C-isotope composition of POM in arable fields had a faster turnover rate than in grassland fields. The woody biomass of hedges presents a much higher C:N ratio than intensive grassland species, such as *Lolium perenne* (Biffi et al., 2023). Therefore, litter under hedges is likely less readily available to decomposition and may persist longer as l-POM than in grassland (Rahman et al., 2013; Liang et al., 2017). However, the strong increase in l-POM and l-POC also illustrates why hedge-associated SOC stocks are rapidly lost after hedge removal. Biffi et al. (2022) showed that SOC stocks beneath young hedges planted over historical hedge boundaries did not significantly differ to adjacent fields. Similarly, a case study in Belgium showed that an additional 25.1 Mg C ha^{-1} stored in the top 23 cm of soil beneath hedges was rapidly lost after their removal (Van Den Berge et al., 2021). The disruption in C inputs from fresh plant biomass and fine root exudates is likely responsible for this rapid loss of SOC stock. Thus, appropriate hedge management is crucial to maintaining SOC stocks. For example, hedge laying, which is a traditional management practice that involves removing part of the hedge aboveground biomass and laying the main shrub stems (i.e. pleachers) horizontally on the ground, entwining them, should be used to favour plant rejuvenation and maintain healthy hedges (Staley et al., 2015). This type of practice also allows increased wood density and sustains aboveground biomass C sequestration over time (Biffi et al., 2023).

5. Conclusions

Hedge planting has been widely encouraged in recent years to contribute towards climate change mitigation and provide multiple benefits to farmed landscapes. Ambitious hedge planting goals have been set in the EU and in the UK to contribute towards net-zero by 2050, and understanding the factors that affect the capacity of soil under hedges to store and sequester SOC is key for their inclusion in greenhouse gas removal balances. This study considered the potential of hedges to store SOC under a range of pedoclimatic conditions across England. We found that the additional SOC stock beneath hedges was not significantly different among locations and that, on average, the hedges stored ~40 Mg C ha^{-1} more than adjacent grassland fields. We also found that net SOC sequestration rates beneath hedges decline rapidly as they grow and mature. Our results show that between two and five decades after hedge planting, soil under hedge woody vegetation cover reaches a new equilibrium, in which the carbon input from plant residues and root exudates does not exceed the carbon lost through mineralization, and that this equilibrium is consistent across UK pedoclimatic conditions. We also found that most of the sequestered SOC is stored in the labile SOM fraction, suggesting that l-POC can be used as an indicator of short-term changes in total SOC stock after planting woody species in agricultural landscapes. Thus, our findings show that, within UK pedoclimatic conditions, the additional SOC stock under hedges is controlled by the presence of woody plant cover and the return of litter and turnover of root biomass associated with hedges, rather than soil properties or climatic factors. Importantly, this also highlights the role of maintaining existing hedges through regular management for SOC storage, as the removal and deterioration of hedges will result in the rapid loss of their associated l-POC.

CRediT authorship contribution statement

Sofia Biffi: Conceptualization, Data curation, Formal analysis, Visualisation, Writing – original draft preparation, Writing – review & editing. **Pippa J. Chapman:** Conceptualization, Methodology, Writing – review & editing, Funding acquisition, Project administration. **Richard P. Grayson:** Methodology. **Joseph Holden:** Funding acquisition. **Jonathan R. Leake:** Funding acquisition. **Holly Armitage:** Investigation. **Sarah F.P. Hunt:** Investigation. **Guy Ziv:** Conceptualization, Writing – review & editing, Funding acquisition.

Declaration of Competing Interest

The authors declare that there are no relevant financial or non-financial competing interests.

Data availability

The data that support the findings of this study are openly available in Zenodo at https://doi.org/10.5281/zenodo.14197326.

Acknowledgements

This project has been supported with funding from the Global Food Security's 'Resilience of the UK Food System Programme' with support from BBSRC, ESRC, NERC and Scottish Government, as part of the Resilient Dairy Landscapes project (grant BB/R005664/1); the Research England's Quality-related Research Strategic Priorities Funding (QR SPF); and the Natural Environment Research Council as part of the U.K. Soil Security Programme, as part of the SoilBioHedge Project (grant NE/M017044/1 and NE/M017079/1). We would like to thank the Game and Wildlife Conservation Trust for helping to find sampling site in Sussex, and Fraser Hugill for help with finding sites in North Yorkshire. Thanks to all farmers and landowners for allowing us to access their land. Thank you to Josiah Judson for assisting with soil sampling, and to Rachel Gasior and David Ashley for help with laboratory analyses.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2025.109471.

References

Aciego Pietri, J.C., Brookes, P.C., 2008. Relationships between soil pH and microbial properties in a UK arable soil. Soil Biol. Biochem. 40, 1856–1861. https://doi.org/10.1016/j.soilbio.2008.03.020.

Antony, D., Collins, C.D., Clark, J.M., Sizmur, T., 2022. Soil organic matter storage in temperate lowland arable, grassland and woodland topsoil and subsoil. Soil Use Manag.

Arevalo, C.B.M., Bhatti, J.S., Chang, S.X., Sidders, D., 2009. Ecosystem carbon stocks and distribution under different land-uses in north central Alberta, Canada. For. Ecol. Manag. 257, 1776–1785. https://doi.org/10.1016/j.foreco.2009.01.034.

Austin, A.T., Ballaré, C.L., 2010. Dual role of lignin in plant litter decomposition in terrestrial ecosystems. Proc. Natl. Acad. Sci. USA 107, 4618–4622. https://doi.org/10.1073/pnas.0909396107.

Axe, M.S., Grange, I.D., Conway, J.S., 2017. Carbon storage in hedge biomass—a case study of actively managed hedges in England. Agric. Ecosyst. Environ. 250, 81–88. https://doi.org/10.1016/j.agee.2017.08.008.

Bai, Y., Cotrufo, M.F., 2022. Grassland soil carbon sequestration: current understanding, challenges, and solutions. Science 377, 603–608. https://doi.org/10.1126/science.abo2380.

Bailey, V.L., Pries, C.H., Lajtha, K., 2019. What do we know about soil carbon destabilization? Environ. Res. Lett. 14, 083004. https://doi.org/10.1088/1748-9326/ab2c11.

Barr, C.J., Gillespie, M.K., 2000. Estimating hedgerow length and pattern characteristics in Great Britain using Countryside Survey data. J. Environ. Manag. 60, 23–32.

Baudry, J., Bunce, R.G.H., Burel, F., 2000. Hedgerows: an international perspective on their origin, function and management. J. Environ. Manag. 60, 7–22. https://doi.org/10.1006/jema.2000.0358.

Beck, H.E., Zimmermann, N.E., McVicar, T.R., Vergopolan, N., Berg, A., Wood, E.F., 2018. Present and future Köppen-Geiger climate classification maps at 1-km resolution. Sci. Data 5, 180214. https://doi.org/10.1038/sdata.2018.214.

Beka, S., Burgess, P.J., Corstanje, R., Stoate, C., 2022. Spatial modelling approach and accounting method affects soil carbon estimates and derived farm-scale carbon payments. Sci. Total Environ. 827, 154164. https://doi.org/10.1016/j.scitotenv.2022.154164.

Biffi, S., Chapman, P.J., Engler, J.O., Kunin, W.E., Ziv, G., 2024. Using automated passive acoustic monitoring to measure changes in bird and bat vocal activity around hedgerows of different ages. Biological Conservation 296, 110722.

Biffi, S., Chapman, P.J., Grayson, R.P., Ziv, G., 2022. Soil carbon sequestration potential of planting hedgerows in agricultural landscapes. J. Environ. Manag. 307, 114484. https://doi.org/10.1016/j.jenvman.2022.114484.

Biffi, S., Chapman, P.J., Grayson, R.P., Ziv, G., 2023. Planting hedgerows: biomass carbon sequestration and contribution towards net-zero targets. Sci. Total Environ. 892, 164482. https://doi.org/10.1016/j.scitotenv.2023.164482.

Black, K., Lanigan, G., Ward, M., Kavanagh, I., hUallacháin, D.Ó., Sullivan, L.O., 2023. Biomass carbon stocks and stock changes in managed hedgerows. Sci. Total Environ. 871, 162073. https://doi.org/10.1016/j.scitotenv.2023.162073.

Boinot, S., Alignier, A., Pétillon, J., Ridel, A., Aviron, S., 2023. Hedgerows are more multifunctional in preserved bocage landscapes. Ecol. Indic. 154, 110689. https://doi.org/10.1016/j.ecolind.2023.110689.

Bossio, D.A., Cook-Patton, S.C., Ellis, P.W., Fargione, J., Sanderman, J., Smith, P., Wood, S., Zomer, R.J., vonUnger, M., Emmer, I.M., Griscom, B.W., 2020. The role of soil carbon in natural climate solutions. Nat. Sustain. 3, 391–398. https://doi.org/10.1038/s41893-020-0491-z.

Bradley, R., Milne, R., Bell, J., Lilly, A., Jordan, C., Higgins, A., 2005. A soil carbon and land use database for the United Kingdom. Soil Use Manag. 21, 363–369. https://doi.org/10.1079/SUM2005351.

Bramble, D.S.E., Ulrich, S., Schöning, I., Mikutta, R., Brandt, L., Poll, C., Kandeler, E., Mikutta, C., Konrad, A., Siemens, J., Yang, Y., Polle, A., Schall, P., Ammer, C., Kaiser, K., Schrumpf, M., 2023. Formation of mineral-associated organic matter in temperate soils is primarily controlled by mineral type and modified by land use and management intensity. Glob. Change Biol. n/a, e17024. https://doi.org/10.1111/gcb.17024.

Burgess, P., Graves, A., 2022.The potential contribution of agroforestry to net zero objectives. Report for the Woodland Trust. Technical Report, Cranfield University, Bedfordshire:.

Cambardella, C.A., Elliott, E.T., 1992. Particulate soil organic-matter changes across a grassland cultivation sequence. Soil Sci. Soc. Am. J. 56, 777–783. https://doi.org/10.2136/sssaj1992.03615995005600030017x.

Cardinael, R., Chevallier, T., Cambou, A., Béral, C., Barthès, B.G., Dupraz, C., Durand, C., Kouakoua, E., Chenu, C., 2017. Increased soil organic carbon stocks under agroforestry: a survey of six different sites in France. Agric. Ecosyst. Environ. 236, 243–255. https://doi.org/10.1016/j.agee.2016.12.011.

Cardinael, R., Guenet, B., Chevallier, T., Dupraz, C., Cozzi, T., Chenu, C., 2018. High organic inputs explain shallow and deep SOC storage in a long-term agroforestry system – combining experimental and modeling approaches. Biogeosciences 15, 297–317. https://doi.org/10.5194/bg-15-297-2018.

Carvalhais, N., Forkel, M., Khomik, M., Bellarby, J., Jung, M., Migliavacca, M., Mu, M., Saatchi, S., Santoro, M., Thurner, M., Weber, U., Ahrens, B., Beer, C., Cescatti, A., Randerson, J.T., Reichstein, M., 2014. Global covariation of carbon turnover times with climate in terrestrial ecosystems. Nature 514, 213–217. https://doi.org/10.1038/nature13731.

Castellano, M.J., Mueller, K.E., Olk, D.C., Sawyer, J.E., Six, J., 2015. Integrating plant litter quality, soil organic matter stabilization, and the carbon saturation concept. Glob. Change Biol. 21, 3200–3209. https://doi.org/10.1111/gcb.12982.

Chiartas, J.L., Jackson, L.E., Long, R.F., Margenot, A.J., O'Geen, A.T., 2022. Hedgerows on crop field edges increase soil carbon to a depth of 1 meter. Sustainability 14, 12901. https://doi.org/10.3390/su141912901.

Climate Change Committee, 2018.Land use: reducing emissions and preparing for climate change. Technical Report, Climate Change Committee, London.

Cordeiro, C.F.d.S., Rodrigues, D.R., da Silva, G.F., Echer, F.R., Calonego, J.C., 2022. Soil organic carbon stock is improved by cover crops in a tropical sandy soil. Agron. J. 114, 1546–1556.

Cotrufo, M.F., Ranalli, M.G., Haddix, M.L., Six, J., Lugato, E., 2019. Soil carbon storage informed by particulate and mineral-associated organic matter. Nat. Geosci. 12, 989–994. https://doi.org/10.1038/s41561-019-0484-6.

Creamer, C.A., Foster, A.L., Lawrence, C., McFarland, J., Schulz, M., Waldrop, M.P., 2019. Mineralogy dictates the initial mechanism of microbial necromass association. Geochim. Cosmochim. Acta 260, 161–176. https://doi.org/10.1016/j.gca.2019.06.028.

Crossland, M., 2015. The carbon sequestration potential of hedges managed for woodfuel. TWECOM Project Growing Local Energy 41.

De Deyn, G.B., Cornelissen, J.H.C., Bardgett, R.D., 2008. Plant functional traits and soil carbon sequestration in contrasting biomes. Ecol. Lett. 11, 516–531. https://doi.org/10.1111/j.1461-0248.2008.01164.x.

De Stefano, A., Jacobson, M.G., 2018. Soil carbon sequestration in agroforestry systems: a meta-analysis. Agrofor. Syst. 92, 285–299. https://doi.org/10.1007/s10457-017-0147-9.

DEFRA, 2021.The England trees action plan 2021–2024. Technical Report PB 14667, Department for Environment, Food and Rural Affairs, London.

DeGryze, S., Six, J., Paustian, K., Morris, S.J., Paul, E.A., Merckx, R., 2004. Soil organic carbon pool changes following land-use conversions. Glob. Change Biol. 10, 1120–1132. https://doi.org/10.1111/j.1529-8817.2003.00786.x.

Drexler, S., Don, A., 2024. Carbon sequestration potential in hedgerow soils: results from 23 sites in Germany. Geoderma 445, 116878. https://doi.org/10.1016/j.geoderma.2024.116878.

Drexler, S., Gensior, A., Don, A., 2021. Carbon sequestration in hedgerow biomass and soil in the temperate climate zone. Reg. Environ. Change 21, 74. https://doi.org/10.1007/s10113-021-01798-8.

Drexler, S., Thiessen, E., Don, A., 2023. Carbon storage in old hedgerows: the importance of below-ground biomass. GCB Bioenergy n/a. https://doi.org/10.1111/gcbb.13112.

Eze, S., Magilton, M., Magnone, D., Varga, S., Gould, I., Mercer, T.G., Goddard, M.R., 2023. Meta-analysis of global soil data identifies robust indicators for short-term changes in soil organic carbon stock following land use change. Sci. Total Environ. 860, 160484. https://doi.org/10.1016/j.scitotenv.2022.160484.

Eze, S., Palmer, S.M., Chapman, P.J., 2018a. Soil organic carbon stock and fractional distribution in upland grasslands. Geoderma 314, 175–183. https://doi.org/10.1016/j.geoderma.2017.11.017.

Eze, S., Palmer, S.M., Chapman, P.J., 2018b. Soil organic carbon stock in grasslands: effects of inorganic fertilizers, liming and grazing in different climate settings. J. Environ. Manag. 223, 74–84. https://doi.org/10.1016/j.jenvman.2018.06.013.

Follain, S., Walter, C., Legout, A., Lemercier, B., Dutin, G., 2007. Induced effects of hedgerow networks on soil organic carbon storage within an agricultural landscape. Geoderma 142, 80–95. https://doi.org/10.1016/j.geoderma.2007.08.002.

Ford, H., Healey, J.R., Webb, B., Pagella, T.F., Smith, A.R., 2019. How do hedgerows influence soil organic carbon stock in livestock-grazed pasture? Soil Use Manag. 35, 576–584. https://doi.org/10.1111/sum.12517.

Fowler, A.F., Basso, B., Millar, N., Brinton, W.F., 2023. A simple soil mass correction for a more accurate determination of soil carbon stock changes. Sci. Rep. 13, 2242. https://doi.org/10.1038/s41598-023-29289-2.

Fujisaki, K., Chevallier, T., Chapuis-Lardy, L., Albrecht, A., Razafimbelo, T., Masse, D., Ndour, Y.B., Chotte, J.L., 2018. Soil carbon stock changes in tropical croplands are mainly driven by carbon inputs: a synthesis. Agric. Ecosyst. Environ. 259, 147–158. https://doi.org/10.1016/j.agee.2017.12.008.

Gelardi, D.L., Rath, D., Kruger, C.E., 2023. Grounding United States policies and programs in soil carbon science: strengths, limitations, and opportunities. Front. Sustain. Food Syst. 7.

Gifford, R.M., Roderick, M.L., 2003. Soil carbon stocks and bulk density: spatial or cumulative mass coordinates as a basis of expression? Glob. Change Biol. 9, 1507–1514. https://doi.org/10.1046/j.1365-2486.2003.00677.x.

Godbold, D.L., Hoosbeek, M.R., Lukac, M., Cotrufo, M.F., Janssens, I.A., Ceulemans, R., Polle, A., Velthorst, E.J., Scarascia-Mugnozza, G., De Angelis, P., Miglietta, F., Peressotti, A., 2006. Mycorrhizal hyphal turnover as a dominant process for carbon input into soil organic matter. Plant Soil 281, 15–24. https://doi.org/10.1007/s11104-005-3701-6.

Guest, E.J., Palfreeman, L.J., Holden, J., Chapman, P.J., Firbank, L.G., Lappage, M.G., Helgason, T., Leake, J.R., 2022. Soil macroaggregation drives sequestration of organic carbon and nitrogen with three-year grass-clover leys in arable rotations. Sci. Total Environ. 852, 158358. https://doi.org/10.1016/j.scitotenv.2022.158358.

Guillaume, T., Makowski, D., Libohova, Z., Bragazza, L., Sallaku, F., Sinaj, S., 2022. Soil organic carbon saturation in cropland-grassland systems: storage potential and soil quality. Geoderma 406, 115529. https://doi.org/10.1016/j.geoderma.2021.115529.

Guo, Y., Abdalla, M., Espenberg, M., Hastings, A., Hallett, P., Smith, P., 2021. A systematic analysis and review of the impacts of afforestation on soil quality indicators as modified by climate zone, forest type and age. Sci. Total Environ. 757, 143824. https://doi.org/10.1016/j.scitotenv.2020.143824.

Haddix, M.L., Paul, E.A., Cotrufo, M.F., 2016. Dual, differential isotope labeling shows the preferential movement of labile plant constituents into mineral-bonded soil organic matter. Glob. Change Biol. 22, 2301–2312. https://doi.org/10.1111/gcb.13237.

Heckman, K., Hicks Pries, C.E., Lawrence, C.R., Rasmussen, C., Crow, S.E., Hoyt, A.M., von Fromm, S.F., Shi, Z., Stoner, S., McGrath, C., Beem-Miller, J., Berhe, A.A., Blankinship, J.C., Keiluweit, M., Marín-Spiotta, E., Monroe, J.G., Plante, A.F., Schimel, J., Sierra, C.A., Thompson, A., Wagai, R., 2022. Beyond bulk: density fractions explain heterogeneity in global soil carbon abundance and persistence. Glob. Change Biol. 28, 1178–1196. https://doi.org/10.1111/gcb.16023.

Hedges, L.V., Gurevitch, J., Curtis, P.S., 1999. The meta-analysis of response ratios in experimental ecology. Ecology 80, 1150–1156.

Heim, A., Schmidt, M.W.I., 2007. Lignin is preserved in the fine silt fraction of an arable Luvisol. Org. Geochem. 38, 2001–2011. https://doi.org/10.1016/j.orggeochem.2007.08.009.

Hemingway, J.D., Rothman, D.H., Grant, K.E., Rosengard, S.Z., Eglinton, T.I., Derry, L.A., Galy, V.V., 2019. Mineral protection regulates long-term global preservation of natural organic carbon. Nature 570, 228–231. https://doi.org/10.1038/s41586-019-1280-6.

Hernandez-Ramirez, G., Sauer, T.J., Cambardella, C.A., Brandle, J.R., James, D.E., 2011. Carbon sources and dynamics in afforested and cultivated corn belt soils. Soil Sci. Soc. Am. J. 75, 216–225. https://doi.org/10.2136/sssaj2010.0114.

Hewins, D.B., Lyseng, M.P., Schoderbek, D.F., Alexander, M., Willms, W.D., Carlyle, C.N., Chang, S.X., Bork, E.W., 2018. Grazing and climate effects on soil organic carbon concentration and particle-size association in northern grasslands. Sci. Rep. 8, 1336. https://doi.org/10.1038/s41598-018-19785-1.

Hoffland, E., Kuyper, T.W., Comans, R.N.J., Creamer, R.E., 2020. Eco-functionality of organic matter in soils. Plant Soil 455, 1–22. https://doi.org/10.1007/s11104-020-04651-9.

Holden, J., Grayson, R.P., Berdeni, D., Bird, S., Chapman, P.J., Edmondson, J.L., Firbank, L.G., Helgason, T., Hodson, M.E., Hunt, S.F.P., Jones, D.T., Lappage, M.G., Marshall-Harries, E., Nelson, M., Prendergast-Miller, M., Shaw, H., Wade, R.N., Leake, J.R., 2019. The role of hedgerows in soil functioning within agricultural landscapes. Agric. Ecosyst. Environ. 273, 1–12. https://doi.org/10.1016/j.agee.2018.11.027.

IPCC, 2006.IPCC guidelines for national greenhouse gas inventories - agriculture, forestry and other land use. Technical Report Volume 4, Intergovernmental Panel on Climate Change.

Jarmain, C., Cummins, T., Jovani-Sancho, A.J., Nairn, T., Premrov, A., Reidy, B., Renou-Wilson, F., Tobin, B., Walz, K., Wilson, D., Byrne, K.A., 2023. Soil organic carbon stocks by soil group for afforested soils in Ireland. Geoderma Reg. 32, e00615. https://doi.org/10.1016/j.geodrs.2023.e00615.

Jobbágy, E.G., Jackson, R.B., 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecol. Appl. 10, 423–436. https://doi.org/10.1890/1051-0761(2000)010[0423:TVDOSO]2.0.CO;2.

Laganière, J., Angers, D.A., Paré, D., 2010. Carbon accumulation in agricultural soils after afforestation: a meta-analysis. Glob. Change Biol. 16, 439–453. https://doi.org/10.1111/j.1365-2486.2009.01930.x.

Lal, R., 2018. Digging deeper: a holistic perspective of factors affecting soil organic carbon sequestration in agroecosystems. Glob. Change Biol. 24, 3285–3301. https://doi.org/10.1111/gcb.14054.

Lavallee, J.M., Soong, J.L., Cotrufo, M.F., 2020. Conceptualizing soil organic matter into particulate and mineral-associated forms to address global change in the 21st century. Glob. Change Biol. 26, 261–273. https://doi.org/10.1111/gcb.14859.

Lesaint, L., Viaud, V., Menasseri-Aubry, S., 2023. Influence of soil properties and land use on organic carbon storage in agricultural soils near hedges. Soil Use Manag. 39, 1140–1154.

Levin, G., Angelidis, I., Gyldenk, S., 2020.Assessment of change in biomass from 2006 to 2014/2015 of non-forest woody vegetation in Denmark. Technical Report Technical Report No. 178, Aarhus University, DCE – DanishCentre for Environment and Energy.

Liang, X., Yuan, J., Yang, E., Meng, J., 2017. Responses of soil organic carbon decomposition and microbial community to the addition of plant residues with different C:N ratio. Eur. J. Soil Biol. 82, 50–55. https://doi.org/10.1016/j.ejsobi.2017.08.005.

Litza, K., Alignier, A., Closset-Kopp, D., Ernoult, A., Mony, C., Osthaus, M., Staley, J., Van Den Berge, S., Vanneste, T., Diekmann, M., 2022. Hedgerows as a habitat for forest plant species in the agricultural landscape of Europe. Agric. Ecosyst. Environ. 326, 107809. https://doi.org/10.1016/j.agee.2021.107809.

Llorente, M., Glaser, B., Turrión, M.B., 2017. Effect of land use change on contents and distribution of monosacharides within density fractions of calcareous soil. Soil Biol. Biochem. 107, 260–268.

Lugato, E., Lavallee, J.M., Haddix, M.L., Panagos, P., Cotrufo, M.F., 2021. Different climate sensitivity of particulate and mineral-associated soil organic matter. Nat. Geosci. 14, 295–300. https://doi.org/10.1038/s41561-021-00744-x.

Luo, Z., Feng, W., Luo, Y., Baldock, J., Wang, E., 2017. Soil organic carbon dynamics jointly controlled by climate, carbon inputs, soil properties and soil carbon fractions. Glob. Change Biol. 23, 4430–4439. https://doi.org/10.1111/gcb.13767.

MAA, 2021.Ministère de l'Agriculture et de la Souveraineté alimentaire: Programme Plantons des haies! ⟨https://agriculture.gouv.fr/francerelance-50-meu-pour-planter-7-000-km-de-haies-en-2-ans⟩.

Mayer, S., Wiesmeier, M., Sakamoto, E., Hübner, R., Cardinael, R., Kühnel, A., Kögel-Knabner, I., 2022. Soil organic carbon sequestration in temperate agroforestry systems – a meta-analysis. Agric. Ecosyst. Environ. 323, 107689. https://doi.org/10.1016/j.agee.2021.107689.

Meier, I.C., Leuschner, C., 2010. Variation of soil and biomass carbon pools in beech forests across a precipitation gradient. Glob. Change Biol. 16, 1035–1045. https://doi.org/10.1111/j.1365-2486.2009.02074.x.

Met Office, 2022a.Met Office MIDAS Open: UK Land Surface Stations Data (1853-current).

Met Office, 2022b.UK Regional Climates. Technical Report.

Minasny, B., Malone, B.P., McBratney, A.B., Angers, D.A., Arrouays, D., Chambers, A., Chaplot, V., Chen, Z.S., Cheng, K., Das, B.S., Field, D.J., Gimona, A., Hedley, C.B., Hong, S.Y., Mandal, B., Marchant, B.P., Martin, M., McConkey, B.G., Mulder, V.L., O'Rourke, S., Richer-de-Forges, A.C., Odeh, I., Padarian, J., Paustian, K., Pan, G., Poggio, L., Savin, I., Stolbovoy, V., Stockmann, U., Sulaeman, Y., Tsui, C.C., VÅgen, T.G., van Wesemael, B., Winowiecki, L., 2017. Soil carbon 4 per mille. Geoderma 292, 59–86. https://doi.org/10.1016/j.geoderma.2017.01.002.

Montgomery, I., Caruso, T., Reid, N., 2020. Hedgerows as ecosystems: service delivery, management, and restoration. Annu. Rev. Ecol. Evol., Syst. 51.

Murphy, B.W., Wilson, B.R., Koen, T., 2019. Mathematical functions to model the depth distribution of soil organic carbon in a range of soils from New South Wales, Australia under different land uses. Soil Syst. 3, 46. https://doi.org/10.3390/soilsystems3030046.

Nair, P.K.R., 2012. Carbon sequestration studies in agroforestry systems: a reality-check. Agrofor. Syst. 86, 243–253. https://doi.org/10.1007/s10457-011-9434-z.

Natural England, 2010.Regional Agricultural Land Classification Maps.

Plaza, C., Courtier-Murias, D., Fernández, J.M., Polo, A., Simpson, A.J., 2013. Physical, chemical, and biochemical mechanisms of soil organic matter stabilization under conservation tillage systems: a central role for microbes and microbial by-products in C sequestration. Soil Biol. Biochem. 57, 124–134. https://doi.org/10.1016/j.soilbio.2012.07.026.

Poeplau, C., 2021. Grassland soil organic carbon stocks along management intensity and warming gradients. Grass Forage Sci. 76, 186–195. https://doi.org/10.1111/gfs.12537.

Poeplau, C., Don, A., 2013. Sensitivity of soil organic carbon stocks and fractions to different land-use changes across Europe. Geoderma 192, 189–201. https://doi.org/10.1016/j.geoderma.2012.08.003.

Poeplau, C., Vos, C., Don, A., 2017. Soil organic carbon stocks are systematically overestimated by misuse of the parameters bulk density and rock fragment content. Soil 3, 61–66. https://doi.org/10.5194/soil-3-61-2017.

R CoreTeam, 2023.R: a Language and Environment for Statistical Computing. R Foundation for Statistical Computing..

Rahman, M.M., Tsukamoto, J., Rahman, M.M., Yoneyama, A., Mostafa, K.M., 2013. Lignin and its effects on litter decomposition in forest ecosystems. Chem. Ecol. 29, 540–553.

Robertson, A.D., Paustian, K., Ogle, S., Wallenstein, M.D., Lugato, E., Cotrufo, M.F., 2019. Unifying soil organic matter formation and persistence frameworks: the MEMS model. Biogeosciences 16, 1225–1248. https://doi.org/10.5194/bg-16-1225-2019.

Robertson, H., Marshall, D., Slingsby, E., Newman, G., 2012.Economic, Biodiversity, Resource Protection and Social Values of Orchards: a Study of Six Orchards by the Herefordshire Orchards Community Evaluation Project.Natural England Commissioned Reports.

Rodrigues, L., Hardy, B., Huyghebeart, B., Fohrafellner, J., Fornara, D., Barančíková, G., Bárcena, T.G., De Boever, M., DiBene, C., Feizienė, D., Kätterer, T., Laszlo, P., O'Sullivan, L., Seitz, D., Leifeld, J., 2021. Achievable agricultural soil carbon sequestration across Europe from country-specific estimates. Glob. Change Biol. 27, 6363–6380. https://doi.org/10.1111/gcb.15897.

Rovira, P., Sauras, T., Salgado, J., Merino, A., 2015. Towards sound comparisons of soil carbon stocks: a proposal based on the cumulative coordinates approach. CATENA 133, 420–431. https://doi.org/10.1016/j.catena.2015.05.020.

Rowell, D.L., 2014. Soil Science: Methods & Applications. Routledge.

RPA, 2022.BN11: Planting New Hedges.

Rumpel, C., Amiraslani, F., Chenu, C., Garcia Cardenas, M., Kaonga, M., Koutika, L.S., Ladha, J., Madari, B., Shirato, Y., Smith, P., Soudi, B., Soussana, J.F., Whitehead, D., Wollenberg, E., 2020. The 4p1000 initiative: opportunities, limitations and challenges for implementing soil organic carbon sequestration as a sustainable development strategy. Ambio 49, 350–360. https://doi.org/10.1007/s13280-019-01165-2.

Schweizer, S.A., Mueller, C.W., Höschen, C., Ivanov, P., Kögel-Knabner, I., 2021. The role of clay content and mineral surface area for soil organic carbon storage in an arable toposequence. Biogeochemistry 156, 401–420. https://doi.org/10.1007/s10533-021-00850-3.

Six, J., Callewaert, P., Lenders, S., De Gryze, S., Morris, S.J., Gregorich, E.G., Paul, E.A., Paustian, K., 2002. Measuring and understanding carbon storage in afforested soils by physical fractionation. Soil Sci. Soc. Am. J. 66, 1981–1987. https://doi.org/10.2136/sssaj2002.1981.

Staley, J.T., Amy, S.R., Adams, N.P., Chapman, R.E., Peyton, J.M., Pywell, R.F., 2015. Restructuring hedges: rejuvenation management can improve the long term quality of hedgerow habitats for wildlife in the UK. Biol. Conserv. 186, 187–196. https://doi.org/10.1016/j.biocon.2015.03.002.

Stewart, C.E., Paustian, K., Conant, R.T., Plante, A.F., Six, J., 2008a. Soil carbon saturation: evaluation and corroboration by long-term incubations. Soil Biol. Biochem. 40, 1741–1750. https://doi.org/10.1016/j.soilbio.2008.02.014.

Stewart, C.E., Plante, A.F., Paustian, K., Conant, R.T., Six, J., 2008b. Soil carbon saturation: linking concept and measurable carbon pools. Soil Sci. Soc. Am. J. 72, 379–392. https://doi.org/10.2136/sssaj2007.0104.

Tan, Z., Lal, R., Owens, L., Izaurralde, R.C., 2007. Distribution of light and heavy fractions of soil organic carbon as related to land use and tillage practice. Soil Tillage Res. 92, 53–59. https://doi.org/10.1016/j.still.2006.01.003.

Terasaki Hart, D.E., Yeo, S., Almaraz, M., Beillouin, D., Cardinael, R., Garcia, E., Kay, S., Lovell, S.T., Rosenstock, T.S., Sprenkle-Hyppolite, S., Stolle, F., Suber, M., Thapa, B., Wood, S., Cook-Patton, S.C., 2023. Priority science can accelerate agroforestry as a natural climate solution. Nat. Clim. Change 13, 1179–1190. https://doi.org/10.1038/s41558-023-01810-5.

Thomas, C., 2022.Woodland and trees in the farmed landscape: towards a diverse, resilient and vibrant agroforestry and farm woodland economy for the UK. Technical Report. Cumulus Consultants Ltd 1 Gainsborough House, Campden Business Park Battle Brook Drive Chipping Campden Gloucestershire GL55 6JX.

Totsche, K.U., Amelung, W., Gerzabek, M.H., Guggenberger, G., Klumpp, E., Knief, C., Lehndorff, E., Mikutta, R., Peth, S., Prechtel, A., Ray, N., Kögel-Knabner, I., 2018. Microaggregates in soils. J. Plant Nutr. Soil Sci. 181, 104–136. https://doi.org/10.1002/jpln.201600451.

Van Den Berge, S., Vangansbeke, P., Baeten, L., Vanneste, T., Vos, F., Verheyen, K., 2021. Soil carbon of hedgerows and 'ghost' hedgerows. Agrofor. Syst. https://doi.org/10.1007/s10457-021-00634-6.

Van Reeuwijk, L.P., 2002. Procedures for Soil Analysis, sixth ed. International soil reference and information Centre (ISRIC), Wageningen.

Vanguelova, E.I., Nisbet, T.R., Moffat, A.J., Broadmeadow, S., Sanders, T.G.M., Morison, J.I.L., 2013. A new evaluation of carbon stocks in British forest soils. Soil Use Manag. 29, 169–181. https://doi.org/10.1111/sum.12025.

Viaud, V., Kunnemann, T., 2021. Additional soil organic carbon stocks in hedgerows in crop-livestock areas of western France. Agric. Ecosyst. Environ. 305, 107174. https://doi.org/10.1016/j.agee.2020.107174.

Vityakon, P., Meepech, S., Cadisch, G., Toomsan, B., 2000. Soil organic matter and nitrogen transformation mediated by plant residues of different qualities in sandy acid upland and paddy soils. NJAS Wagening. J. Life Sci. 48, 75–90. https://doi.org/10.1016/S1573-5214(00)80006-8.

von Haden, A.C., Yang, W.H., DeLucia, E.H., 2020. Soils' dirty little secret: depth-based comparisons can be inadequate for quantifying changes in soil organic carbon and other mineral soil properties. Glob. Change Biol. 26, 3759–3770. https://doi.org/10.1111/gcb.15124.

Vos, C., Don, A., Hobley, E.U., Prietz, R., Heidkamp, A., Freibauer, A., 2019. Factors controlling the variation in organic carbon stocks in agricultural soils of Germany. Eur. J. Soil Sci. 70, 550–564. https://doi.org/10.1111/ejss.12787.

Walter, C., Merot, P., Layer, B., Dutin, G., 2003. The effect of hedgerows on soil organic carbon storage in hillslopes. Soil Use Manag. 19, 201–207. https://doi.org/10.1111/j.1475-2743.2003.tb00305.x.

Ward, S.E., Smart, S.M., Quirk, H., Tallowin, J.R.B., Mortimer, S.R., Shiel, R.S., Wilby, A., Bardgett, R.D., 2016. Legacy effects of grassland management on soil carbon to depth. Glob. Change Biol. 22, 2929–2938. https://doi.org/10.1111/gcb.13246.

Wei, X., Li, X., Jia, X., Shao, M., 2013. Accumulation of soil organic carbon in aggregates after afforestation on abandoned farmland. Biol. Fertil. Soils 49, 637–646. https://doi.org/10.1007/s00374-012-0754-6.

Wendt, J.W., Hauser, S., 2013. An equivalent soil mass procedure for monitoring soil organic carbon in multiple soil layers. Eur. J. Soil Sci. 64, 58–65. https://doi.org/10.1111/ejss.12002.

Wenzel, W.W., Philipsen, F.N., Herold, L., Kingsland-Mengi, A., Laux, M., Golestanifard, A., Strobel, B.W., Duboc, O., 2023. Carbon sequestration potential and fractionation in soils after conversion of cultivated land to hedgerows. Geoderma 435, 116501. https://doi.org/10.1016/j.geoderma.2023.116501.

Witzgall, K., Steiner, F.A., Hesse, B.D., Riveras-Muñoz, N., Rodríguez, V., Teixeira, P.P.C., Li, M., Oses, R., Seguel, O., Seitz, S., Wagner, D., Scholten, T., Buegger, F., Angst, G., Mueller, C.W., 2024. Living and decaying roots as regulators of soil aggregation and organic matter formation—from the rhizosphere to the detritusphere. Soil Biol. Biochem. 197, 109503. https://doi.org/10.1016/j.soilbio.2024.109503.

Woodland Trust, 2021.State of the UK's woods and trees 2021. Technical Report.

Wuest, S.B., 2009. Correction of bulk density and sampling method biases using soil mass per unit area. Soil Sci. Soc. Am. J. 73, 312–316. https://doi.org/10.2136/sssaj2008.0063.

Yan, J., Wang, L., Hu, Y., Tsang, Y.F., Zhang, Y., Wu, J., Fu, X., Sun, Y., 2018. Plant litter composition selects different soil microbial structures and in turn drives different litter decomposition pattern and soil carbon sequestration capability. Geoderma 319, 194–203. https://doi.org/10.1016/j.geoderma.2018.01.009.

Zomer, R.J., Neufeldt, H., Xu, J., Ahrends, A., Bossio, D., Trabucco, A., van Noordwijk, M., Wang, M., 2016. Global tree cover and biomass carbon on agricultural land: the contribution of agroforestry to global and national carbon budgets. Sci. Rep. 6, 29987. https://doi.org/10.1038/srep29987.