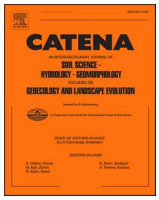
Catena 247 (2024) 108465

Contents lists available at ScienceDirect

Catena

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



Impacts of arable reforestation on soil carbon and nutrients are dependent upon interactions between soil depth and tree species

Josiah B. Judson [a,*], Pippa J. Chapman [a], Joseph Holden [a], Marcelo V. Galdos [b]

[a] School of Geography, University of Leeds, Leeds LS2 9JT, UK
[b] Rothamsted Research, Sustainable Soils and Crops, Harpenden, UK

ARTICLE INFO

Keywords:
Agroforestry
C storage
Soil structure
Land-use change
Litter
Hydrology

ABSTRACT

Recent interest in temperate farm woodland has focussed on strengthening delivery of ecological and economic benefits from land. However, impacts of temperate farm woodland on soil properties and carbon inventories are poorly studied. With field samples and measurements taken at 35-year-old agroforestry experiment we determine how functioning in three components of the soil column (forest floor, topsoil (0–30 cm) and subsoil (>30 cm)) respond to land-use change, tree species choice and small-scale random variability in soil properties. Tree species influenced soil nutrient dynamics in the forest floor and topsoil, with Hazel forest floor material 27 % less concentrated in phosphorus (P) but containing 50 % more soil organic carbon (SOC) stock than Cherry or Sycamore. Change in land use from arable to woodland controlled soil bulk density, organic matter content and C storage in topsoil, with 15 % (11.8 t ha^{-1}) more SOC stock in 0–30 cm soil beneath woodland compared with arable. In subsoil, tree species and land cover influence over soil functioning was insignificant. Notably, no net difference between arable and woodland soil C storage was found when the 0–50 cm part of the profile was considered as a whole, although net C storage was highly variable by plot. 35 years following planting, soil structure and SOC storage were only different in the forest floor and topsoil compared to the adjacent arable system. Each soil component therefore has its own functioning 'signature' in response to afforestation. Future policy support for farm woodland must account for this complexity.

1. Introduction

Competition in temperate landscapes between economic production and ecosystem service delivery has prompted significant interest in reforestation (Ashwood et al., 2019). This is particularly the case on farmland, where agroecological practices such as agroforestry may simultaneously be capable of delivering ecological and economic benefits (Burgess, 1999; Araujo et al., 2012; Torralba et al., 2016; Judson et al., 2023). However, success in benefit delivery from afforestation requires thorough understanding of system interdependencies: between inputs, such as species choice or system design, and outputs such as soil function. These are less well studied in temperate areas: the complexity of tree-soil-atmosphere interactions makes it challenging to isolate specific inputs from other confounding variables, and interdependencies are less well-characterised.

Incorporating trees into farm systems has been cited by bodies such as the IPCC (2014) and UK Climate Change Committee (CCC, 2020a) as an essential component of future climate resilience, and many national (and supra-national) carbon inventories rely on carbon dioxide (CO_2) drawdown in woodland and soils to meet emissions reduction obligations (Smith et al., 2022; UNDESA, 2022). The UK's Climate Change Committee recommends that woodland be incorporated on a minimum of 10 % of farmland by 2050 in order to reach net-zero emissions under their balanced pathway (CCC, 2020b). Yet scaling up farm woodland creation in temperate areas is limited by lack of understanding of benefits, lack of market incentives and cost (Sollen-Norrlin et al., 2020). Directing financial incentives towards best land-use change is always challenging at policy level, and in the case of afforestation, land managers lack contextually specific information on planting design for the successful delivery of multiple benefits. Many temperate studies on farm woodland have focussed on soil C sequestration (De Stefano and Jacobson, 2018; Mayer et al., 2022), with others considering outcomes such as nutrient dynamics (Oelbermann and Voroney, 2007), hydrological functioning (Marshall et al., 2014; Monger et al., 2022) and biodiversity improvements (Varah et al., 2013) or combinations of variables and drivers (Amorim et al., 2022).

* Corresponding author.
E-mail address: gyjbj@leeds.ac.uk (J.B. Judson).

https://doi.org/10.1016/j.catena.2024.108465
Received 31 January 2024; Received in revised form 16 August 2024; Accepted 10 October 2024
Available online 30 October 2024

There has been less focus on specific drivers of soil function in the context of arable land afforestation – such as soil type, tree species, climate, topography, historic management – nor on depths within the soil column at which they have strongest influence. As with many sustainability challenges, delivering ecosystem benefit from farmland soils is a wicked problem in which stakeholders have differing reference frames and objectives from which to judge outcomes (Rittel and Webber, 1973; Bouma et al., 2011). In the case of afforestation, different stakeholders may place emphasis on soil functions in different depth horizons. For example, practitioners requiring nutrient availability for crop production are looking for benefits in topsoil (e.g. Donn et al., 2014), whereas stakeholders interested in landscape soil carbon inventories or flood management need to consider the response of subsoil in addition to shallower horizons to afforestation (e.g. Rogger et al., 2017; De Stefano and Jacobson, 2018).

In light of this we sample a carefully-replicated 35-year-old farm woodland experiment in Yorkshire, UK to consider the response of three soil depth horizons – forest floor material, topsoil (0–30 cm) and subsoil (>30 cm) – to three drivers of soil function, in order to assess their relevance to different stakeholders. The first driver is land-use change: we consider how different horizons of the soil profile respond to woodland planted on previously arable land, and the effect of land-use change on properties such as soil organic carbon (SOC) stock, hydrological functioning and nutrient availability. The second driver is tree species: we consider how key soil properties are controlled by three broadleaf timber species commonly planted for agroforestry. These are sycamore (*Acer pseudoplanatus* L.), cherry (*Prunus avium* L.) and hazel (*Corylus maxima* Miller, cv. Kentish Cob). The third is small-scale random variability in soil properties (pre-planting) between experimental replicates, whereby we demonstrate the extent to which outcomes are controlled by pre-existing conditions, irrespective of treatment. The aim of the study is to determine how each of the three depth horizons respond to drivers in terms of delivery of ecosystem services from arable soil following afforestation.

2. Methods

2.1. Study site

The University of Leeds research farm is a commercially-run mixed arable and pasture farm 5 km west of Tadcaster, northern England. Mean annual precipitation between 1992 and 2021 was 639 mm (Met Office, 2006) and mean annual temperature was 9.5 °C (Met Office, 2019). The soils in this study are Calcaric Endoleptic Cambisols from the Aberford Series: well-drained calcareous brown earths extensively found on low-dipping Permian and Jurassic limestones in both mid- and northern England (Cranfield University, 2022). Aberford soils in the UK are commonly under arable cultivation with much more limited areas of pasture. Soil depths on the University Farm range between 30 cm and 70 cm depth, broadly in line with depths found elsewhere for the soil series.

An agroforestry experiment was established at the farm in 1988 (Fig. 1). Four replicate plots of approximately 110 × 110 m area were planted close to one another, each containing a silvoarable agroforestry plot and adjacent woodland areas. Each site included a 48 × 30 m arable 'control' area in an adjacent, conventionally cropped field. The arable control areas have been under continuous, intensive management from 1994 and since 2009 have been in a four-year rotation of winter wheat, oilseed rape, potatoes or vining peas, and winter or spring barley. They are ploughed annually and fertilised with 140–150 kg N ha^{-1}, 70–86 kg K ha^{-1}, 23 kg P ha^{-1} and 22 kg S ha^{-1}, in addition to 8 t ha^{-1} pig manure in 2018 (Guest et al., 2022). Today, two of the arable control areas remain, with two (corresponding to plots 1 and 2) unusable having been converted for separate experimental trials. Plot 1 and 2 control areas were therefore replaced with two areas to the west of Plot 1 for this study (Fig. 1).

The woodland areas of the agroforestry experiment were each planted with three furniture timber species – ash (*Fraxinus excelsior* L.), sycamore (*Acer pseudoplanatus* L.) and wild cherry (*Prunus avium* L.) – in

Fig. 1. Map of experimental plots used in this study at University of Leeds research farm showing sampling locations by species (coloured crosses) and plot number (white figures). Sampled woodland areas shown as dotted boxes, with ash areas shown in grey. Upper right inset shows site location.

adjacent 26 × 20 m rectangular blocks. These were established with 130 trees per block at a 2 × 2 m spacing, a density typical of conventional forestry. A fourth species – hazel (*Corylus maxima* Miller, cv. Kentish Cob) – was planted as an orchard in 4 × 4 m spacing. Surrounding each plot and enclosing the silvoarable and forestry areas is a windbreak of 11 cultivars of poplar (*Populus alba* L.) and four of willow (*Salix sp*). The woodland areas represent a unique opportunity for study as they are carefully replicated, subdivided by species and analogous to mature areas of farm woodland which might be found elsewhere in either arable or pastoral contexts.

2.2. Sampling strategy

Soil samples were first collected in May 2022 beneath three of the four species in the woodland areas – hazel, wild cherry and sycamore – and in arable control areas (Fig. 1). Within the woodland areas, ash blocks were omitted from our study as the spread of ash dieback has meant farms are no longer planting ash. Sampling sites beneath each of the three woodland species were replicated across the four plots, producing 12 woodland sample locations in total (Fig. 1). A location was chosen near the centre of each block, at the midpoint of the shortest distance between two adjacent trees. In the case of the hazel orchard this was 2 m from the nearest tree; for the sycamore and wild cherry plots this was 1 m from the nearest tree. Four control sample locations (arable, which were all in winter wheat in 2022) were sampled, with two adjacent to Plot 1 and one adjacent to each of Plots 3 and 4. Control samples were taken within the wheat crop itself, away from tramlines. The entire sampling procedure was repeated in February 2023.

Soil samples were extracted from five depth intervals between 0 and 50 cm. At each sampling location a 5 cm diameter ring corer (Eijkelkamp, Holland) was used to extract intact 100 cm^3 soil cores at 2.5–7.5, 12.5–17.5, 22.5–27.5, 32.5–37.5 and 42.5–47.5 cm (representing 0–10, 10–20, 20–30, 30–40 and 40–50 cm, respectively) below the surface for the determination of bulk density and moisture content. For the remainder of the study, we refer to 'surface soil' as 0–10 cm depth, 'topsoil' as 0–30 cm depth and 'subsoil' as > 30 cm depth. At three woodland sample locations (within plots 1 and 4) limestone bedrock was reached at depths shallower than 50 cm, such that deeper layers could not be sampled. Separate, loose, soil samples from the same sample locations and depth intervals were collected for determination of SOC, nitrogen (N) and phosphorus (P) concentrations.

L (litter), F (fermented) and H (humic) organic horizons (hereafter 'forest floor') were sampled from the forest floor at each of the woodland sample locations. A 50 cm^2 quadrat was laid at the midpoint of the shortest distance between two trees within each of the woodland blocks. From the quadrat, the L horizon was collected as loose leaf litter and twigs and placed in plastic bags. Removal of the litter exposed the F and H horizons beneath. The F and H horizon was subsequently sampled using a single 5 cm diameter (100 cm^3) bulk density ring gently hammered into the underlying soil surface, with contents retained intact and returned to the laboratory for analysis.

Tension infiltrometers were used in February 2023 to measure infiltration at a constant tension of −2 cm. Eight measurements were taken for each tree species and arable control across all plots. Infiltration estimates were combined with constants for clay loam soil derived from the method of van Genuchten (van Genuchten and Nielsen, 1985) to estimate near-surface saturated hydraulic conductivity (K_s). This was carried out at the same distance to tree used for soil sample locations across the woodland and arable sites.

A space-for-time substitution approach was used throughout, in which a point-in-time soil sample from beneath woodland species was compared against arable soil samples, with arable areas assumed to represent baseline ($t = 0$) state before afforestation. This widely used approach (Cardinael et al., 2015; Biffi et al., 2022) assumes that the field and woodland sites were equivalent prior to land-use change – a reasonable assumption given the woodland plots were situated in the corner of arable fields.

2.3. Laboratory analysis

2.3.1. Bulk density and organic matter

Soil samples taken intact with the 100 cm^3 ring corer were weighed and subsequently oven dried at 105 °C for 12 h before being weighed again for the determination of bulk density. Moisture content was determined for each of these samples by comparing soil mass before and after oven drying at 105 °C. Roots and stones were extracted and weighed in order to correct for their presence in the soil. Bulk density was calculated by subtracting the root and stone fraction from the final mass. Finally, oven-dry samples were heated to 550 °C for 12 h in order to determine soil organic matter content (%) by the loss on ignition method. These were subsequently weighed, with loss on ignition determined as the change in mass between 105 °C (oven-drying) and 550 °C (ignition), divided by oven-dry mass.

Loose, field-moist L horizon material from the forest floor was weighed and oven dried at 65 °C for five days before being weighed again to determine moisture content. Field-moist ring samples of combined F and H (hereafter 'FH') horizons and underlying mineral soil were returned intact to the laboratory. Colour change was used to determine the boundary between F and H horizons and underlying mineral soil, and combined FH thickness was measured and recorded using electronic Vernier callipers. F and H horizons were combined due to difficulty in distinguishing and separating them. For each sample, the FH horizons were cut off horizontally at the measured depth and weighed, before being oven dried at 105 °C for 12 h and then reweighed to determine moisture content. Uncorrected bulk density of the FH-horizon was determined using oven-dry mass in combination with the ring diameter and measured thickness of the FH horizon.

2.3.2. Soil nutrients and pH

Plant-available N and Olsen's phosphorous (Olsen, 1954) were determined using field-moist soil samples from surface soil (0–10 cm). Loose samples were returned to the laboratory and immediately homogenised by passing through a 5 mm sieve. For determination of N concentration, approximately 10 g field-moist sample was combined with 50 mL 1 M KCl solution and shaken for 1 h at 150 cycles min^{-1} using a shaker table. These were subsequently passed through Whatman 42 filter paper into centrifuge tubes, with NO_3-N and NH_4-N content determined using a Skalar San ++ (Skalar Analytical B.V., Netherlands) continuous flow auto-analyser. Following available N determination, remaining field-moist sample was dried at room temperature and passed through a 2 mm sieve. Approximately 2.5 g air-dried soil was weighed into a shaker bottle, combined with 50 mL of 0.5 M $NaHCO_3$ solution and mixed for 1 h at 150 cycles min^{-1} using a shaker table. These were subsequently passed through Whatman 42 filter paper into centrifuge tubes, with Olsen's phosphorous (PO_4-P) content determined using the same continuous flow auto-analyser.

For L- and FH-horizon samples determination of Olsen's phosphorous was not possible and total P was instead determined using a digestion method. Approximately 0.4 g of air-dried and shredded (L) or < 0.5 mm sieved (FH) sample was combined with 4.4 mL digestion reagent consisting of 30 % hydrogen peroxide (87.5 mL), selenium (0.105 g), sulphuric acid (105 mL) and lithium sulphate (3.5 g). Digestion of plant material was carried out at slowly increasing temperature up to 300 °C until the digest became colourless, with total P in the extract determined colorimetrically using a Skalar San ++ continuous flow auto-analyser.

Following nutrient determinations, soil pH was measured using the <2 mm fraction of air-dry soil. Approximately 20 g air dry soil was combined with 40 mL deionised water and stirred for 15 min. The pH was measured in the deionised water only, before being measured again after the addition of 250 μL $CaCl_2$.

2.3.3. Total C, total N and SOC

Total N content of mineral soil and FH samples was determined using loose samples dried at room temperature. Following nutrient determination, dried samples were ground to <150 μm using a Retsch MM400 ball mill (RETSCH GmbH, Germany). Approximately 4 mg of <150 μm sample was weighed using a six-figure balance into tin capsules, crushed into a small cube to remove air from the sample, and subsequently introduced into an Elemental Vario EL cube (Elementar Analysensysteme GmbH, Germany) combustion analyser to determine concentration of total carbon and nitrogen. To determine organic C content of mineral soil and FH samples the procedure was repeated, with the exception that all samples were weighed into silver capsules and 30 μL of 15 % HCl added to each sample to remove carbonates. Samples were left to react before being oven dried for 2 h at 80 °C and analysed for SOC content with the combustion analyser. For L-horizon samples the above process was repeated, with the exception that plant material dried at 65 °C was shredded to <500 μm using a Retsch SM100 cutting mill (RETSCH GmbH, Germany) before being introduced into capsules for the combustion analyser. C stock was calculated as the product of measured SOC content and dry soil/forest floor material mass per unit area.

2.4. Mineral soil mass corrections

Comparing soil properties to fixed depths following land-use change, such as afforestation of previously arable land, can lead to over- or underestimations of ratio-based soil properties (von Haden et al., 2020). All ratio-based soil measurements were therefore normalised according to a reference quantity of mineral soil in order to correct for land-use change effects on bulk density and SOM. For this purpose, the aggregated control area samples were used as they are assumed to represent $t = 0$ under space-for-time substitution. Mineral soil mass was calculated as the mass of dry soil per unit area in the aggregated control area samples to five reference depths, corresponding to sample depths used in the study (0–10 cm, 0–20 cm, 0–30 cm, 0–40 cm and 0–50 cm). For a given soil property in woodland treatments (e.g., moisture, SOC), cumulative mass of the property was calculated to the same reference depths and plotted against cumulative mineral soil mass. A monotonically-increasing cubic spline function (von Haden et al., 2020) was used to fit these data, from which corrected (or equivalent soil mass − ESM) values were interpolated using cumulative mineral soil values from the reference (Control) areas. Using this procedure, ESM corrected values were calculated for SOC (concentration and stocks), total N, SOC:N, PO_4-P, NO_3-N and total plant-available N (NO_3-N + NH_4-N).

2.5. Statistical analysis

Data used for calculation of bulk density, SOC stocks, SOC/N ratio and moisture were tested for normality (Shapiro-Wilk) and homoscedasticity (Bartlett) in order to meet assumptions for ANOVA and pairwise Tukey tests. Where assumptions were not met, a non-parametric Kruskal-Wallis test was used in place of ANOVA, followed by pairwise Dunn's tests with a Bonferroni correction. All tests were undertaken using *SciPy* (Virtanen et al., 2020) and *statsmodels* (Seabold and Perktold, 2010) within the Python environment (v. 3.10). For land cover comparisons between woodland and arable treatments, Tukey-Kramer mean comparisons were used (owing to unequal sample sizes), with statistical significance evaluated at $p < 0.05$. Comparison between tree species treatments was undertaken using ANOVA mean comparison tests, with significant differences between individual species treatments evaluated using post-hoc pairwise Tukey mean comparisons.

Following land-cover and species treatment comparison, the significance of land cover, species and random plot variability as driving effects was determined for soil functioning indicators. A linear mixed model approach was considered for evaluation of species vs plot variability effect strengths, with plot variability treated as a random effect. However this model was not feasible as the Control treatment is no longer replicated with the same plot areas as Cherry, Hazel and Sycamore treatments. Thus, species and plot variability effects were compared using two-way factorial ANOVA which considers interactions between factors as well as their individual contributions. This was initially undertaken with individual species and plot effects and an interaction term combining them, after which, if the interaction term was not significant at $p < 0.05$, the test was repeated without the interaction term. The land cover effect could not be included in factorial ANOVA due to the absence of control (treeless) data within species and plot categories – thus pre-determined Tukey-Kramer p-values were used to determine significance of the land cover effect on soil function indicators. Although this method does not permit *relative* comparison of fixed and random effect strengths on soil properties, the constraint is imposed by changes to the study site design since planting, and we can nonetheless determine which effect is making significant contribution to variance in soil properties at any given depth.

3. Results

3.1. Bulk density, SOM and hydraulic conductivity

Mean bulk density of arable soil (1.46 ± 0.04 g cm^{-3}, $n = 8$) was nearly 20 % higher than for woodland soil (1.28 ± 0.02 g cm^{-3}, $n = 24$) at 0–10 cm depth (Fig. 2a, Supp. Table 2, $p < 0.001$). This was accompanied by a significant difference in SOM between woodland (8.12 ± 0.23 %) and arable (6.82 ± 0.62 %) soils at 0–10 cm ($p = 0.022$). At all other depths there were no significant differences in bulk density between arable and woodland soils. Surface (0–10 cm) soil bulk density was not significantly different between each of the tree species treatments (Fig. 2b, Cherry 1.29 ± 0.02 g cm^{-3}, Hazel 1.29 ± 0.03 g cm^{-3}, Sycamore 1.26 ± 0.03 g cm^{-3}, $n = 8$, $p = 0.672$).

Surface soil K_s varied significantly between woodland and arable treatments (Supp. Table 3). Mean K_s was ~ 2.5 × faster (3.24 mm hr^{-1}, 2.64 – 3.99) in arable soil than woodland soil (1.31 mm hr^{-1}, 1.15–1.49) ($p = 0.003$). No significant differences in surface K_s were observed between tree species ($p = 0.498$).

3.2. Soil organic carbon

In the top 10 cm, SOC content was significantly higher in woodland plots (3.05 ± 0.11 %) than in arable areas (1.91 ± 0.06 %, $p < 0.001$) (Fig. 3a, Supp. Table 2). In deeper layers, however, the difference was reversed. Between 20 and 30 cm depth, soil in arable fields had higher SOC content (1.73 ± 0.06 %) compared with woodland (1.48 ± 0.05 %, $p = 0.010$), and at 30–40 cm depth the difference was even larger (arable 1.33 ± 0.08 %, woodland 0.90 ± 0.05 %, $p < 0.001$). No significant differences in SOC content were observed between any of the individual tree species treatments at any depth interval (Fig. 3b), nor were there any significant differences in the SOC content of either the litter or FH horizon of the forest floor beneath each of the tree species treatments.

Differences in SOC stock between arable and woodland treatments were significant in topsoil (0–30 cm). This difference was most pronounced at 0–10 cm depth ($p < 0.001$), with a significant difference in SOC stock also observed at 0–30 cm ($p = 0.027$), but not at 0–50 cm ($p = 0.944$, Fig. 4a, Supp. Table 2). Between 0 and 10 cm depth, 62 % more SOC stock was found in soil beneath trees (45.3 ± 1.7 t ha^{-1}) compared with soil beneath arable control plots (27.9 ± 1.0 t ha^{-1}) ($p < 0.001$). Including SOC stock from LFH horizons in the woodland total increased the woodland mean SOC stock to 54.3 ± 1.9 t ha^{-1}, a difference of 26.4 t C ha^{-1} (+95 %) compared with arable plots.

Differences in SOC stock between individual species treatments were not significant at 0–10 cm. However, C stock within the combined LFH horizons varied significantly between species treatments. Although the total LFH biomass beneath the three species was not significantly different between species, there was significantly more C stored in the

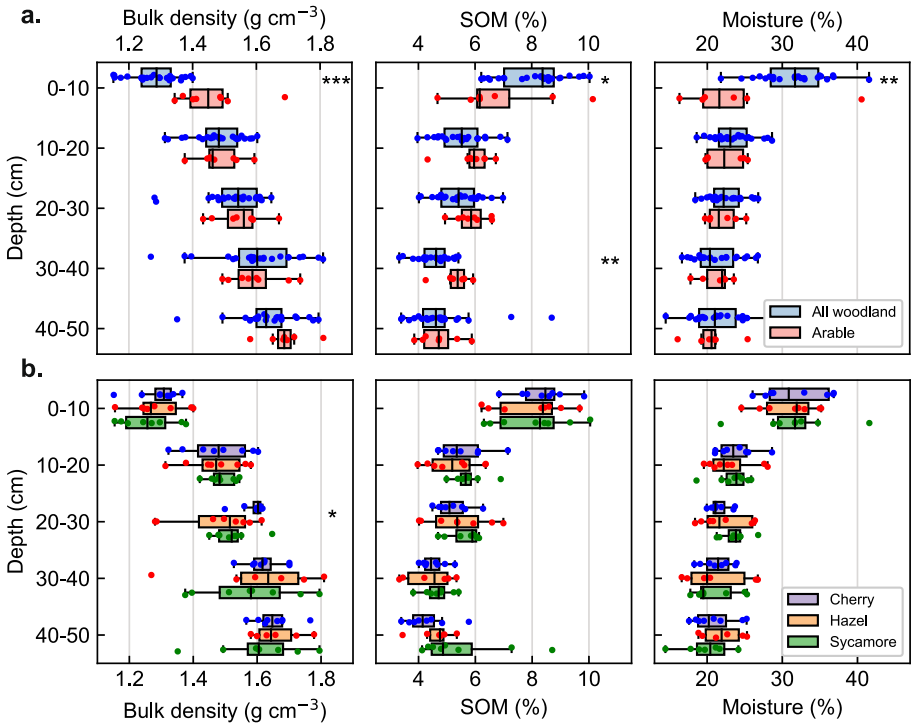


Fig. 2. Variation in bulk density, soil organic matter (SOM) and moisture with increasing depth for a. afforested plots (n = 8) and arable control plots (n = 8) and b. Cherry (n = 8), Hazel (n = 8) and Sycamore (n = 8). Coloured dots show actual sample values. Asterisks denote significant differences between values at the same depth (* – $p < 0.05$, ** – $p < 0.01$, *** – $p < 0.001$).

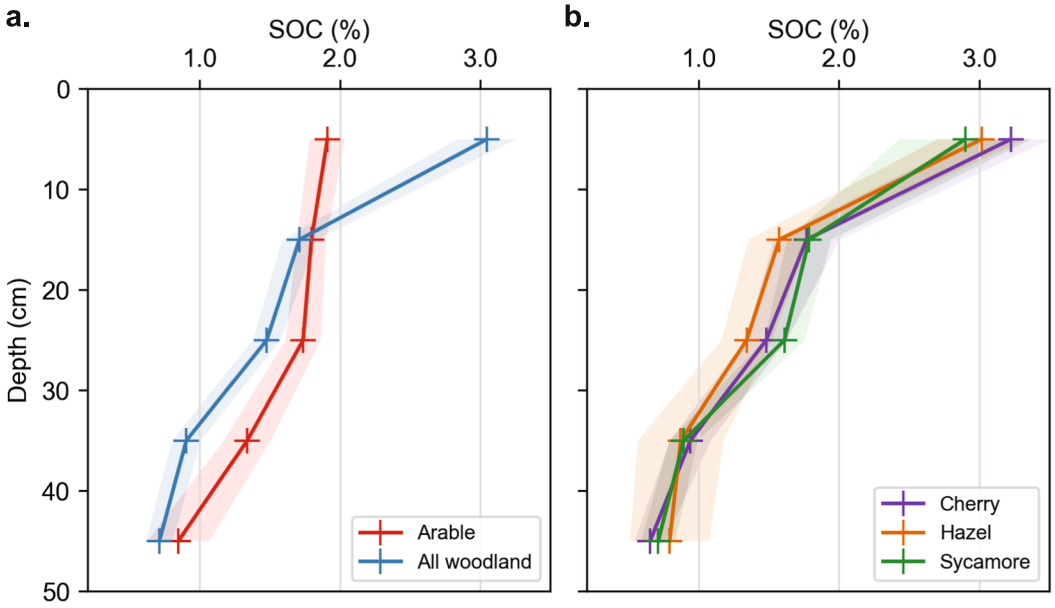


Fig. 3. SOC concentration with depth for a. all woodland and arable (land cover, left) and b. individual tree species (right). Shaded areas show 95% confidence intervals.

LFH horizons beneath Hazel (11.66 ± 1.26 t ha^{-1}) compared with either Cherry (7.86 ± 0.59 t ha^{-1}) or Sycamore (7.61 ± 0.57 t ha^{-1}) (p = 0.016) (Fig. 4b).

The whole topsoil (0–30 cm) of the afforested plots contained + 15 %, or 11.8 t C ha^{-1}, more SOC stock (93.3 ± 2.8 t ha^{-1}) compared with arable controls (81.5 ± 2.6 t ha^{-1}) (p = 0.027). Including the LFH horizons in the SOC total increased the topsoil woodland mean SOC stock to 102.4 ± 2.9 t ha^{-1}), a 20.9 t C ha^{-1} (+26 %) difference compared with arable (Fig. 5) ($p < 0.001$). As observed at 0–10 cm depth, differences in 0–30 cm SOC stock between individual tree species were not significant for the topsoil.

When considering the total soil profile depth (0–50 cm), there was no significant difference in SOC stock between woodland (119 ± 4 t ha^{-1}) and arable (118 ± 5 t ha^{-1}) treatments (p = 0.944). This remained the

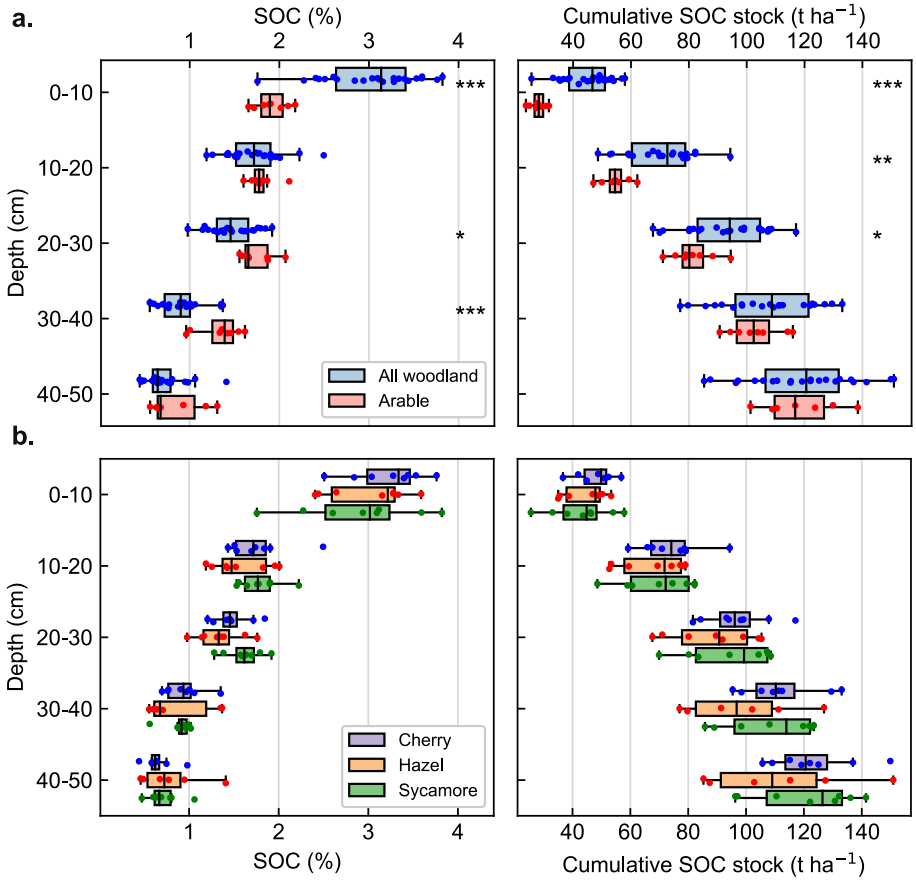


Fig. 4. Variation in soil organic carbon (SOC) as concentration and cumulative stock with increasing depth for a. afforested plots ($n = 24$) and arable control plots ($n = 8$) and b. Cherry ($n = 8$), Hazel ($n = 8$) and Sycamore ($n = 8$). Coloured dots show actual sample values. Asterisks denote significant differences between values at each depth (*: $p < 0.05$, **: $p < 0.01$, ***: $p < 0.001$).

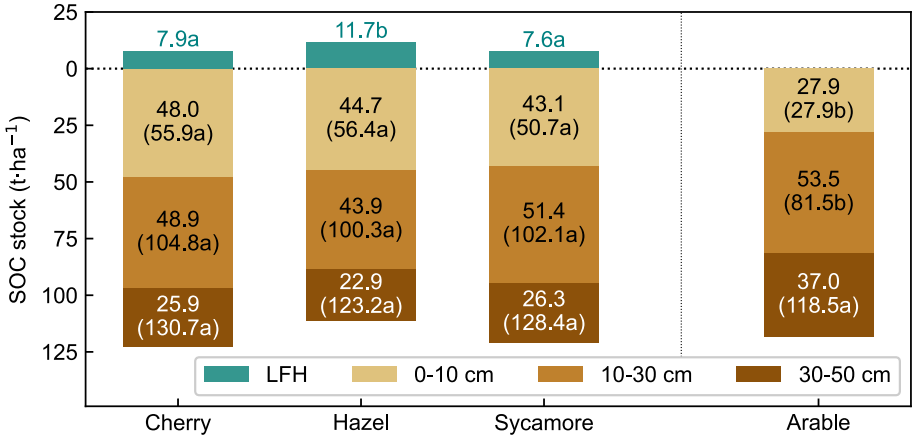


Fig. 5. SOC stock by depth interval (LFH (tree litter and humus), 0–10 cm, 10–30 cm, 30–50 cm) for Cherry, Hazel, Sycamore and Arable treatments. SOC stock in t·ha^{-1} shown for each depth interval, with cumulative total (including LFH) shown in brackets. Letters denote within-depth significant differences between cumulative totals.

case even when the SOC stock in LFH horizons were included in the woodland SOC total ($p = 0.251$), even though LFH C increased woodland C stock by 7.3 % to 128 ± 4 t ha^{-1}.

3.3. Soil nutrients and pH

The top 10 cm of arable soil contained more nitrate-N (NO_3-N) (15.7 ± 4.2 mg kg^{-1}, $p = 0.011$) and phosphate-P (PO_4-P) (32.3 ± 3.0 mg kg^{-1}, $p = 0.002$) compared with woodland soil (NO_3-N: 4.1 ± 1.0 mg kg^{-1}, PO_4-P: 15.7 ± 2.4 mg kg^{-1}). Differences in nutrient content were also observed between species treatments in both LFH horizons and mineral soil (Fig. 6, Supp. Table 1). Total P in the L horizon was significantly different ($p = 0.011$) between species, with Hazel containing less total P (0.95 ± 0.06 g kg^{-1}) than litter beneath either Cherry (1.33 ± 0.06 g kg^{-1}) or Sycamore (1.26 ± 0.09 g kg^{-1}) (Fig. 6a). Total P in the combined FH horizons was not significantly different between

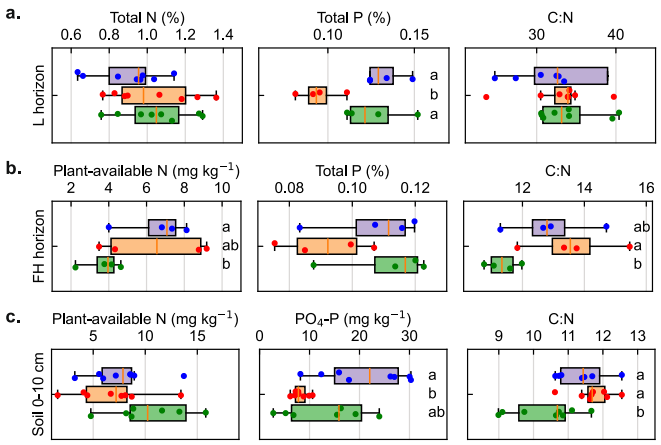


Fig. 6. Nutrient concentrations and C:N for a. L-horizon, b. FH combined horizon and c. 0–10 cm soil between individual species treatments. Letters denote significant differences within horizons. Boxes coloured by tree species (orange – Hazel, purple – Cherry, green – Sycamore).

species, although rank order matched L horizon interspecies differences (Fig. 6b). These differences had the same rank order (Fig. 6c) and were weakly correlated (Supp. Fig. 1, $p = 0.101$) with interspecies variation in surface soil PO_4-P. At 0–10 cm soil depth, soil beneath Hazel contained less than half of the PO_4-P (8.2 ± 0.5 mg kg^{-1}, $n = 8$) than soil beneath Cherry (21.0 ± 3.0 mg kg^{-1}, $n = 8$) ($p = 0.002$). Sycamore surface soil PO_4-P (17.9 ± 5.7 mg kg^{-1}, $n = 8$) was more variable meaning differences with other species were not observed.

Total N was less variable than total P or PO_4-P between species treatments in both LFH and surface soil (0–10 cm) horizons. Interspecies total N differences in L (Fig. 6a) or FH (Fig. 6b) horizons were not observed at $p < 0.05$. However in the FH combined horizon, Cherry (6.57 ± 0.90 mg kg^1) and Hazel (6.44 ± 1.48 mg kg^{-1}) contained 1.75 times more plant-available N than Sycamore (3.70 ± 0.52 mg kg^{-1}) (Supp. Table 1). Surface soil interspecies differences in NO_3-N were not significant ($p = 0.319$), while surface soil plant-available N ($NO_3 + NH_4$) showed greater but still non-significant ($p = 0.084$) variability (Fig. 6c).

The C:N ratio at 0–10 cm depth was significantly higher in woodland soil (11.25 ± 0.23) than arable soil (9.95 ± 0.36, $p = 0.006$). Between species, 0–10 cm Sycamore soil had a significantly lower C:N ratio (10.4 ± 0.3) than either Cherry (11.7 ± 0.4) or Hazel (11.7 ± 0.2) (Fig. 6c, Supp. Table 2, $p = 0.013$). This corresponded with near-significant ($p = 0.075$) C:N differences in the FH horizons, with Sycamore FH having lower C:N (11.4 ± 0.3) than Cherry (12.9 ± 0.7) or Hazel (13.6 ± 0.7).

Soil pH was not significantly variable in surface soil between arable and woodland treatments (in H_2O: $p = 0.094$; in $CaCl_2$: $p = 0.173$), nor between individual species treatments (in H_2O: $p = 0.689$; in $CaCl_2$: 0.823, Supp. Table 3).

3.4. Relative impact of land cover, tree species and plot variability

The significance of each driving effect (land cover, tree species, and plot natural variability) on soil properties varied with depth (Table 1). Tree species had greatest impact on C and nutrient content of LFH horizons, whereas land cover accounted for variability in SOC stock, bulk density and hydrological functioning in the topsoil. In the subsoil, bulk density and cumulative SOC stock were most strongly controlled by spatial differences between plots.

3.4.1. LFH horizons

Tree species exerted a significant influence on the P concentration of the L-horizon ($p = 0.044$). In contrast, the L-horizon N concentration was not influenced by species ($p = 0.290$), although underlying FH-horizon plant-available N was controlled by tree species ($p = 0.031$). Plot variability had no effect on L horizon total P ($p = 0.950$) or N ($p = 0.389$) stock, although FH-horizon plant-available N was significantly different between plots ($p = 0.029$). Species differences were the dominant control on LFH total SOC stock ($p = 0.012$), with no plot variability effect ($p = 0.158$). Land cover effects were not considered for LFH horizons due to the absence of LFH horizons in arable areas.

3.4.2. Topsoil

Land cover was a significant control on surface soil (0–10 cm) bulk density ($p < 0.001$), saturated hydraulic conductivity (K_s) ($p = 0.001$) and SOM ($p = 0.022$). Cumulative SOC stock in surface soil was also primarily controlled by land cover ($p < 0.001$). However, plot variability also significantly controlled K_s ($p < 0.001$) and SOM ($p = 0.020$), and SOC stock was controlled by plot variability ($p = 0.040$) when organic C from LFH horizons was included in the total.

Species differences exerted some control over surface soil nutrient stocks. PO_4-P differences were significantly controlled by species ($p = 0.050$), although species control over plant-available N was only significant at 90 % confidence ($p = 0.075$). It was not possible to compare species nutrient effects with the influence of land cover due to the use of synthetic fertiliser on the arable land. No direct species control was observed on soil structural variables or SOC stock/concentration in surface soil. Surface soil C:N was the only indicator observed which was controlled by land cover ($p = 0.006$), species differences ($p = 0.001$) and plot variability ($p = 0.004$).

Table 1
Influence of land cover, plot variability and species effects on soil variables. Values are two-way ANOVA (plot variability, species) and Tukey-Kramer (land cover) p-values. Values shown in bold are significant at $p < 0.05$. Dashes indicate combinations which were either not sampled, or in the case of the cross terms (species*plot), not significant (after which ANOVA was rerun without cross term).

Sample type	*Effect*	pH	BD	N	P	SOC	C:N	Ks
Litter (LFH)	plot variability	–	–	**0.029**	†0.950	0.158	0.912	–
	species	–	–	**0.031**	†**0.044**	**0.012**	0.159	–
	species*plot	–	–	–	–	–	–	–
Shallow soil (0–10 cm)	land cover	0.094	**<0.001**	*0.023	*0.002	**<0.001**	**0.006**	**0.001**
	plot variability	0.069	0.341	0.128	0.181	0.095	**0.004**	**<0.001**
	species	0.499	0.666	0.075	**0.050**	0.435	**0.001**	0.415
	species*plot	–	–	–	–	–	–	**0.040**
Deep soil (40–50 cm)	land cover	–	0.197	–	–	0.944	**0.038**	–
	plot variability	–	**0.027**	–	–	**0.002**	0.537	–
	species	–	0.263	–	–	0.499	0.776	–
	species*plot	–	–	–	–	–	–	–

* Land-cover p-values for N and P greyed out due to influence of synthetic fertiliser on arable areas.
† P corresponds to L-horizon total P for LFH values, PO_4-P for soil measurements.

3.4.3. Subsoil

No land-cover effect on cumulative SOC stock was observed below 30 cm depth. At 50 cm, plot variability was the dominant control on both bulk density ($p = 0.027$) and cumulative SOC stock ($p = 0.002$). Notably, bedrock was reached at shallower depth in plots 1 and 4, potentially affecting compaction deeper in the soil profile. SOM concentration varied significantly between plots for all depth intervals between 0 and 40 cm (0–10 cm $p = 0.020$, 30–40 cm $p = 0.001$).

4. Discussion

Having presented differences in soil properties 35 years following afforestation of arable soil with three broadleaf species, we discuss implications of these differences for soil functioning at three different depth horizons, and how the contextual drivers of land cover, species and small scale random variability control ecosystem service delivery at each depth. We summarise these effects and discuss how each component of the soil profile beneath woodland has its own signature in terms of which contextual controls are in operation, and on which soil outcomes (Fig. 7). It must be noted that not all depths were sampled for all soil properties and drivers (e.g. bulk density or K_s of forest floor, subsoil nutrient dynamics) and differences in management (fertiliser application in arable areas) limit some comparisons. Moreover, relative magnitude of driving effects was not discernible due to pre-existing experimental design and statistical constraints. However, this does not preclude drawing out useful high-level findings. We conclude by discussing implications, including the extent to which farm woodland can support climate resilience in future temperate landscapes.

4.1. Forest floor functioning

Forest floor functioning was predominantly controlled by tree species (Fig. 7), which exerted significant control over chemical composition of litter material. This has implications for N and P dynamics – for example, significantly less P content was found beneath Hazel, and significantly less available N content found beneath Sycamore (Fig. 6). However, forest floor C storage was also species-controlled, with considerably more C stock beneath Hazel than either Sycamore or Cherry (Fig. 5).

Woodland LFH horizons in this study contributed a mean of 7 % extra OC stock to SOC stored in the soil profile to 50 cm depth. A 7 % mean forest floor contribution to total SOC stock is larger than contributions reported elsewhere. For example, Gao et al. (2014), in a study of two broadleaf and two conifer species of a similar age (~40 years) in a semi-arid temperate region of China, estimated a forest floor contribution of < 1 % of SOC stock. Contributions of forest floor material to C stocks are important for wider-scale evaluation of C storage potential of agroforestry and farm woodland. Despite being less recalcitrant, the risk of forest floor C being removed is minimised as land beneath woodland is much less often disturbed than other agricultural areas.

The quantity of extra C accumulated in forest floor material varied by species, as was also found by Gao et al. (2014). Despite each species having accumulated a similar amount of LFH biomass over 35 years, LFH beneath Hazel contained ~ 1.5 times more C stock than either Sycamore or Cherry, although there was no difference in SOC storage between the species over the whole soil column (Fig. 5). Species choice must therefore be considered in estimating forest floor contributions to C storage as an ecosystem service, particularly in the case of species monocultures, and more work is needed to characterise interspecies differences in litter quality beyond the three under study.

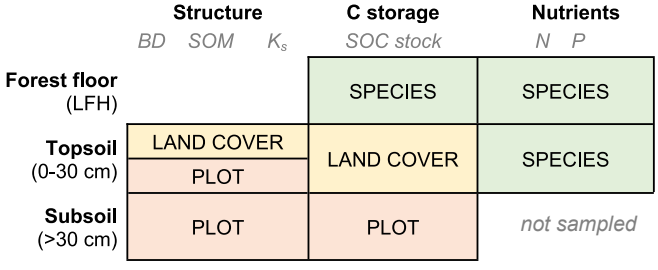


Fig. 7. Illustrative diagram of dominant high-level contextual controls (*land cover* – yellow, *tree species* – green, *plot variability* – red) on soil functioning by depth horizon and soil property type.

Species differences in forest floor C and N content have implications for decomposition and transfer of litter C into surface soil. The C:N ratio of forest floor material controls its decomposition rate, however unlike studies with more sites/species for comparison (e.g. Cools et al., 2014), C:N was not significantly variable between species in our study. We did not find coupling between intraspecific litter C and surface soil OC variability, with surface soil SOC stock not controlled by tree species.

The N and P content of forest floor material was controlled by tree species (Table 1, Fig. 7) and was observed to have weak control on surface soil nutrient dynamics. Hazel litter contained less total P compared with Sycamore or Cherry, matching interspecies variability in surface soil PO_4-P, with 0–10 cm soil beneath Hazel containing less than half the PO4-P of Sycamore or Cherry (Fig. 6). Litter and surface soil N concentration was more weakly species-controlled and coupled, yet plant-available N was species-controlled in the forest floor. Other studies have reported the influence of tree species on N, P and C dynamics. For example, Hoosbeek et al. (2018) found variable influence of tree species in a Nicaraguan silvopastoral system over N and P stocks in the topsoil: Jícaro (*Crescentia alata*) promoted greater N stock and Guácimo (*Guazuma ulmifolia*) promoted P stock. Similarly, Amorim et al. (2022) found surface soil (0–15 cm) beneath red oak (*Quercus rubra L.*) and pecan (*Carya illinoinensis* (Wangenh.) K. Koch.) to have significantly different C:N ratio and SOC concentration, which they attribute to distinctive leaf litter and nutrient inputs. Forest floor functioning and surface soil nutrient availability and fertility are thus closely linked with tree species choice. Afforestation is a key component of maintaining sustainable nutrient availability in agricultural soils, and species choice will depend on fertility needs of soils under management.

4.2. Topsoil functioning

Tree species effects were weaker in topsoil than the forest floor, and functioning was most strongly determined by land cover change. Topsoil has significant potential to promote C storage following land cover change from arable to woodland, with considerably more C stored beneath woodland at 0–10 cm (+17.4 t ha^{-1}, +0.50 t ha^{-1} year^{-1}), and a significant, but less pronounced, difference at 0–30 cm (+11.8 t C ha^{-}1, +0.34 t ha^{-1} year^{-1}, Fig. 4a). C is transferred to surface (and deeper) soil horizons from root and shoot litter and also root exudates (Jobbágy et al., 2001; Haichar et al., 2014), all of which are incorporated at higher rates in areas such as woodland where plant diversity is greater (Eisenhauer et al., 2017; Judson et al., 2023). Topsoil C storage was, however, insensitive to tree species (Fig. 4b) and there was no significant difference between plots.

Many other studies have demonstrated potential for extra C storage in afforested arable topsoils over time (Dawson and Smith, 2007; Upson et al., 2016; De Stefano and Jacobson, 2018; Ashwood et al., 2019; Mayer et al., 2022), mainly reporting differences at 0–20 cm (Table 2). Ashwood et al. (2019) found that differences in topsoil SOC stock were greatest when land was converted from arable to woodland, with pasture having similar surface SOC stock to woodland. The difference of +16.0 t C ha^{-1} (+0.46 t C ha^{1} year^{1}) that we measured between arable soils and young woodland (35 year old) at 0–20 cm depth is smaller than that found by Ashwood et al. (2019) (+37.2 t C ha^{1}; +0.74 t C ha^{-1} year^{-1}) for the same depth interval, although it is larger than results found elsewhere for 0–20 cm SOC change following agriculture to agroforestry conversion (Table 2). Ashwood et al. (2019) included OC

Table 2
Comparison of changes in SOC stock and sequestration rates between this study and studies of similar tree age, land-use change and climate zone. A 0–20 cm depth interval is used to aid comparison.

Study	Location	Age	LUC	Change in SOC stock					
				LFH/OH		0–20 cm		0–40 cm	
				Amount (t ha^{-1})	Rate (t ha^{-1} yr^{-1})	Amount (t ha^{-1})	Rate (t ha^{-1} yr^{-1})	Amount (t ha^{-1})	Rate (t ha^{-1} yr^{-1})
This study	Yorks, UK	35	Arable to woodland	+9.0	+0.26	+16.0	+0.46	Not significant	
Mayer et al. (2022)	Various temperate	28	Various*	–	–	+7.0	+0.21	+10.1	+0.36
Cardinael et al. (2015)	Montpellier, France	18	Arable to silvoarable†	–	–	+17.0	+0.94	+17.7	+0.98
Ashwood et al. (2019)	Midlands, UK	50	Arable to woodland	+9.1	+0.18	+37.2	+0.74	+63.5	+1.27
Upson and Burgess (2013)	Beds, UK	20	Arable to silvoarable†	–	–	+7.2	+0.36	+22.0	+1.10

* Mayer et al. (2022) is a meta-analysis which includes the following land-use change (LUC) types: pasture to silvopasture, arable to silvoarable or hedge.
† Figures shown are for samples at similar distance-to-tree as in our study.

stock values for the partially decomposed OH litter layer above the soil surface, which contributed + 9.1 t C ha^{-1} (+0.18 t C ha^{1} year^{1}), a very similar figure to the contribution from LFH in our study (+9.0 t C ha^{-1}; +0.26 t C ha^{1} year^{1}). Assuming the arable control had the same land-use history as the afforested plots prior to tree planting, woodland plots in our study contained 24.9 t C ha^{1} (+46 %) extra OC in 0–20 cm topsoil and overlying LFH compared with arable fields.

However, changes in SOC stock were very sensitive to depth. In contrast with other studies (Table 2), for the 0–40 cm depth interval (extending into the subsoil) we found no additional C stock resulting from afforestation even with LFH OC included, effectively cancelling out benefits measured in topsoil alone. This demonstrates the importance of sampling beneath topsoil to determine the holistic effect of land-use change on C storage, and the potential for soil to offset CO_2 emissions.

Topsoil nutrient dynamics were weakly controlled by species in the uppermost 10 cm with significant variability in PO_4-P content (Table 1), and, as mentioned above, this is likely to be the result of coupling with overlying forest floor material. However, it is difficult to contrast this with other drivers such as land cover due to the influence of synthetic fertiliser application in arable areas on nutrient dynamics. Determining afforestation control on nutrient dynamics would require further work with control and woodland areas under equivalent management, which we were not able to test.

Topsoil physical and hydrological properties were also determined by land cover change, with some small-scale plot variability. Surface soils under trees were significantly less compacted than soils under continuous arable cultivation, with bulk density of soil in the top 10 cm beneath woodland 17 % lower than in arable areas. This is significant for topsoil functioning and agricultural productivity, as water and nutrient uptake by roots are inhibited in more compacted soils. The decrease in bulk density was driven by the significant increase in SOM, derived from organic matter input from leaf litter and root turnover. Litter input by woodland is a significant driver of surface bulk density changes. Topsoil bulk density (0–20 cm) beneath the 'young' category of trees examined by Ashwood et al. (2019) showed a decrease of similar magnitude to that in our study. They also found that the reduction in bulk density (at 0–20 cm depth) between arable and young secondary (50 – 60 y) woodland was also substantially greater than that observed between pasture and woodland. Similarly aged trees to those in our study, established on pasture in the UK, produced a smaller but significant reduction in bulk density (Upson et al., 2016), although hedges planted in pastures in northern England (Biffi et al., 2022) produced a similar change in bulk density (−17 %) to this study, although for a greater depth (0–30 cm).

Topsoil *Ks* was controlled by land cover, and also showed considerable variability between plots (Table 1). However, in contrast with other studies (Marshall et al., 2014; Holden et al., 2019) it was found to be 2.5 times faster in arable areas than beneath woodland, implying better runoff mitigation in cultivated areas. It would be beneficial to test this finding using infiltration measurements at a range of hydraulic tensions to determine contributions of different pore size classes to overall saturated hydraulic conductivity, as the presence of cracks in drier arable surface soil may be contributing to the difference (Holden and Gell, 2009).

4.3. Subsoil functioning

In subsoil, the influence of land-cover change on soil functioning was minimal. Instead, soil functioning below 30 cm was controlled by pre-existing random variations in soil properties between plots (Table 1, Fig. 7).

Lack of land cover control on subsoil C storage differs substantially from other studies. Upson and Burgess (2013) found that the 20–40 cm depth interval contributed double the SOC (14.8 t ha^{-1}) compared with 0–20 cm depth (7.2 t ha^{-1}) 20 years after planting. Mayer et al. (2022) found in their meta-analysis that 80 % of studies observed an increase in SOC stock between 20 and 40 cm depth over a mean of 28 years, compared with a 70 % increase in the top 20 cm of mineral soil. Only Cardinael et al. (2015) recorded a very small SOC contribution (0.7 t ha^{-1} over 18 years) between 20 and 40 cm depth. In our study, no significant difference in total SOC stock was observed under woodland compared to arable at 0–50 cm depth. In contrast, the SOC stock at 0–50 cm depth was strongly controlled by random plot differences. Sampling only at Plot 4 would imply a significant 22.2 t ha^{-1} (+18.7 %) increase in woodland SOC stock compared with arable areas over 35 years, a sequestration rate of 0.63 t ha^{-1} year^{-1}; whereas sampling only at Plot 2 would imply a significant 18.1 t ha^{-1} (−15.3 %; −0.51 t C ha^{1} year^{-1}) loss over the same time period.

We therefore found SOC storage over the whole (0–50 cm) soil profile to vary considerably more across the study site than as a direct result of tree planting. This implies that afforestation in some locations may not influence overall soil C stocks at all, and further highlights the importance of sampling to greater depth in evaluating C storage potential of afforestation. Differences between our result and those of similar studies mentioned (Table 2) may have a number of explanations. The lower SOC content beneath trees below the arable plough layer (>30 cm) found in this study potentially implies a SOC redistribution effect, in which lack of tillage and soil turnover beneath woodland accounts for significantly higher C stock in topsoil, but not subsoil. As found by Sun et al. (2011), ceasing tillage generates significant SOC increase in the topsoil but not the whole soil profile. Sampling across the whole soil profile reveals the extent to which extra C has been fixed by woodland, or whether it is simply no longer being redistributed as a result of normal arable management.

Contributions of living roots to below-ground OC stocks, which we did not estimate, would be valuable further work, may also account for discrepancies in sub-soil C stocks. Upson et al. (2016) noted the importance of herbaceous vegetation contributions to overall below-ground C beneath new farm woodland, and Drexler et al. (2024) showed in a study on hedgerows that 85 % of below ground biomass C is stored in coarse roots (> 2 mm), which we did not quantify in our study.

New tree roots may also lead to a priming effect, in which exudates and below ground biomass from tree roots increase decomposition of recalcitrant SOC by fungal and microbial communities in the short term (Fontaine et al., 2007), and this may explain lower SOC concentration and stocks in woodland subsoil when compared with arable soil.

Bulk density was also controlled by plot variability in the deepest (40–50 cm) soil layer. The four plots used in our study (along with arable control areas) are sufficiently close (<200 m) to minimise covariates such as soil changes or topography, and have the same age and layout of woodland. It is therefore likely that with increasing depth, soil functioning is most strongly driven by more complex pre-existing plot conditions, such that land-cover changes including afforestation have an increasingly limited effect on functioning. Proximity to limestone bedrock may also control functioning at depth. This highlights the complexity of determining the influence of a single land cover change on soil functioning throughout the whole soil profile.

5. Conclusions

Thirty five years after the conversion of arable land to woodland, functioning in each of three soil horizons – forest floor material, topsoil (0–30 cm) and subsoil (>30 cm) – has its own signature in terms of ecosystem service provision. Tree species controlled forest floor nutrient dynamics and C storage through mediation of chemical content of litter material. In topsoil below the forest floor, species influence was weaker, and functioning was most strongly controlled by land cover change to woodland, with +15 % or 11.8 t ha^{-1} extra C stored compared with arable land. Yet, afforestation was only capable of promoting C storage in forest floor material and topsoil. Extra C storage higher up the soil profile was cancelled out by C deficit below 30 cm, such that there was no significant net difference in C storage between soil under woodland or arable crops. Instead, C storage, bulk density and hydrological properties in subsoil were most strongly determined by small scale random variability in soil properties (pre-planting) between experimental replicates.

These findings firstly demonstrate the complexity of determining ecosystem service delivery from land use change. Considering options in linear terms using single inputs (e.g. woodland, agroforestry) and outputs (e.g. C storage) is insufficient for policy evaluation of carbon inventories and future landscape sustainability. Further consideration of C dynamics between forest floor material and soil, and other interactions between woodland C pools is required to predict the potential long-term ecosystem benefits.

Secondly, variation of benefit delivery within the soil profile demonstrates the need to evaluate findings across the whole soil column. Current schemes paying land managers per hectare for land use changes, such as agroforestry, must both consider whole soil column functioning, and be more rigorously evaluated in terms of interdependencies between factors such as tree species choice, and where beneficial outcomes are delivered. Other inputs such as soil type or management, which would usefully be the subject of further work, are also likely to have a substantial influence.

Thirdly, at greater depth in the soil column, we find benefits to be highly influenced by pre-existing spatial variability in soil properties, even over short (<200 m) distances and with the same soil type and planting design. We note the limitations this implies for tradeable ecosystem 'credit' schemes such as soil C codes, which homogenise benefits over much larger distances and variation in soil or topography.

Finally, soil outcomes from afforestation may trade-off against one another. Although woodland in this study did not appear to increase net soil C storage to 50 cm over 35 years, it was capable of promoting surface soil nutrient addition in the vicinity of trees (in the absence of synthetic inputs) within a shorter, multi-decadal timeframe, and was responsible for transferring significant organic matter to surface soil, lowering bulk density with implications for water storage and flood resilience. Assessment of benefit therefore depends on stakeholder objectives for planting. Notably we did not account for above-ground biomass C stock in woodland totals, which would likely increase C storage benefit.

Future support for farm woodland creation must be context-specific in order for land to deliver both economic and ecological benefits. Moreover, rewards for good practice must respect the diverse range of goals for planting, and how these are borne out in different parts of the soil profile. Combining these objectives will increase the potential for woodland to promote sustainable change in the landscape.

CRediT authorship contribution statement

Josiah B. Judson: Writing – review & editing, Writing – original draft, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Pippa J. Chapman:** Writing – review & editing, Supervision, Methodology, Conceptualization. **Joseph Holden:** Writing – review & editing, Supervision, Methodology, Conceptualization. **Marcelo V. Galdos:** Writing – review & editing, Supervision, Methodology, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

Funding: This work was supported by the Leeds-York-Hull Natural Environment Research Council (NERC) Doctoral Training Partnership (DTP) Panorama [grant number NE/S007458/1], with support from the Woodland Trust as CASE partner. MG acknowledges funding from Rothamsted Research's Growing Health Institute Strategic Programme [BB/X010953/1; BBS/E/RH/230003B]. The authors thank Isobel Lloyd, Francesca Darvill and Jyoti Narsude for assistance with sample collection and field measurements; David Pilbeam for insight and experience at the study site; Holly Armitage, Joshua Greenwood, David Ashley and Rachel Gasior for assistance with sample processing and laboratory analysis; and three anonymous reviewers for comments which have significantly strengthened the manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.catena.2024.108465.

Data availability

Data associated with this paper are available from the University of Leeds at https://doi.org/10.5518/1595.

References

Amorim, H.C.S., Ashworth, A.J., Zinn, Y.L., Sauer, T.J., 2022. Soil organic carbon and nutrients affected by tree species and poultry litter in a 17-year agroforestry site. Agronomy 12.

Araujo, A.S.F., et al., 2012. Microbiological process in agroforestry systems. a review. Agron. Sustain. Dev. 32, 215–226.

Ashwood, F., et al., 2019. Woodland restoration on agricultural land: long-term impacts on soil quality. Restor. Ecol. 27, 1381–1392.

Biffi, S., Chapman, P.J., Grayson, R.P., Ziv, G., 2022. Soil carbon sequestration potential of planting hedgerows in agricultural landscapes. J. Environ. Manage. 307, 114484.

Bouma, J., Van Altvorst, A., Eweg, R., Smeets, P., Van Latesteijn, H., 2011. The role of knowledge when studying innovation and the associated wicked sustainability problems in agriculture. Adv. Agron. 113, 293–323.

Burgess, P., 1999. Effects of agroforestry on farm biodiversity in the UK. Scott. For. 53, 24–27.

Cardinael, R., et al., 2015. Impact of alley cropping agroforestry on stocks, forms and spatial distribution of soil organic carbon - a case study in a Mediterranean context. Geoderma 259–260, 288–299.

CCC, 2020a. Land use: Policies for a Net Zero UK.
CCC, 2020b. The Sixth Carbon Budget: The UK's path to Net Zero.
Cools, N., Vesterdal, L., De Vos, B., Vanguelova, E., Hansen, K., 2014. Tree species is the major factor explaining C: N ratios in European forest soils. For. Ecol. Manage. 311, 3–16.
Cranfield University, 2022. The soils guide. Cranfield University, UK.
Dawson, J.J.C., Smith, P., 2007. Carbon losses from soil and its consequences for land-use management. Sci. Total Environ. 382, 165–190.
De Stefano, A., Jacobson, M.G., 2018. Soil carbon sequestration in agroforestry systems: a meta-analysis. Agrofor. Syst. 92, 285–299.
Donn, S., Wheatley, R.E., McKenzie, B.M., Loades, K.W., Hallett, P.D., 2014. Improved soil fertility from compost amendment increases root growth and reinforcement of surface soil on slopes. Ecol. Eng. 71, 458–465.
Drexler, S., Thiessen, E., Don, A., 2024. Carbon storage in old hedgerows: the importance of below-ground biomass. GCB Bioenergy 16, 1–16.
Eisenhauer, N., et al., 2017. Root biomass and exudates link plant diversity with soil bacterial and fungal biomass. Sci. Rep. 7, 1–8.
Fontaine, S., et al., 2007. Stability of organic carbon in deep soil layers controlled by fresh carbon supply. Nature 450, 277–280.
Gao, Y., Cheng, J., Ma, Z., Zhao, Y., Su, J., 2014. Carbon storage in biomass, litter, and soil of different plantations in a semiarid temperate region of northwest China. Ann. For. Sci. 71, 427–435.
Guest, E.J., et al., 2022. Soil macroaggregation drives sequestration of organic carbon and nitrogen with three-year grass-clover leys in arable rotations. Sci. Total Environ. 852, 158358.
Haichar, F.E.Z., Santaella, C., Heulin, T., Achouak, W., 2014. Root exudates mediated interactions belowground. Soil Biol. Biochem. 77, 69–80.
Holden, J., et al., 2019. The role of hedgerows in soil functioning within agricultural landscapes. Agric. Ecosyst. Environ. 273, 1–12.
Holden, J., Gell, K.F., 2009. Morphological characterization of solute flow in a brown earth grassland soil with cranefly larvae burrows (leatherjackets). Geoderma 152, 181–186.
Hoosbeek, M.R., Remme, R.P., Rusch, G.M., 2018. Trees enhance soil carbon sequestration and nutrient cycling in a silvopastoral system in south-western Nicaragua. Agrofor. Syst. 92, 263–273.
IPCC, 2014. Agriculture, Forestry and Other Land Use (AFOLU). In: Edenhofer, O., et al. (Eds.). Cambridge University Press, Cambridge, UK; New York, NY, USA, pp. 811–922.
Jobbágy, E.G., Jackson, R.B., Biogeochemistry, S., Mar, N., 2001. The distribution of soil nutrients with depth: global patterns and the imprint of plants published by : springer stable URL : http://www.jstor.org/stable/1469627 references linked references are available on JSTOR for this article : You may need to log i. Biogeochemistry 53, 51-77.
Judson, J.B., Holden, J., Chapman, P., Galdos, M.V., 2023. Trees, hedges, agroforestry and microbial diversity. In: Goss, M.J., Oliver, M. (Eds.), Encyclopaedia of Soils in the Environment (second Edition). Elsevier, pp. 469–479.
Marshall, M.R., et al., 2014. The impact of rural land management changes on soil hydraulic properties and runoff processes: Results from experimental plots in upland UK. Hydrol. Process. 28, 2617–2629.
Mayer, S., et al., 2022. Soil organic carbon sequestration in temperate agroforestry systems – A meta-analysis. Agiculture, Ecosystems, and Environment, 323.
Met Office, 2019. MIDAS Open: UK Land Surface Stations Data (1853-current).
Monger, F., Spracklen, D.V., Kirkby, M.J., Schofield, L., 2022. The impact of semi-natural broadleaf woodland and pasture on soil properties and flood discharge. Hydrol. Process. 36, 1–14.
Oelbermann, M., Voroney, R.P., 2007. Carbon and nitrogen in a temperate agroforestry system: using stable isotopes as a tool to understand soil dynamics. Ecol. Eng. 29, 342–349.
Met Office, 2006. MIDAS: UK Daily Rainfall Data.
Olsen, S.R., 1954. Estimation of available phosphorus in soils by extraction with sodium bicarbonate. US Department of Agriculture.
Rittel, H.W., Webber, M.M., 1973. Dilemmas in a general theory of planning. Policy Sci. 4, 155–169.
Rogger, M., et al., 2017. Land use change impacts on floods at the catchment scale: challenges and opportunities for future research. Water Resour. Res. 5209–5219.
Seabold, S., Perktold, J., 2010. statsmodels: Econometric and statistical modeling with python.
Smith, H.B., Vaughan, N.E., Forster, J., 2022. Long-term national climate strategies bet on forests and soils to reach net-zero. Commun. Earth Environ. 3, 1–12.
Sollen-Norrlin, M., Ghaley, B.B., Rintoul, N.L.J., 2020. Agroforestry benefits and challenges for adoption in Europe and beyond. Sustainability (Switzerland) 12, 1–20.
Sun, B., Hallett, P.D., Caul, S., Daniell, T.J., Hopkins, D.W., 2011. Distribution of soil carbon and microbial biomass in arable soils under different tillage regimes. Plant and Soil 338, 17–25.
Torralba, M., Fagerholm, N., Burgess, P.J., Moreno, G., Plieninger, T., 2016. Do European agroforestry systems enhance biodiversity and ecosystem services? A meta-analysis. Agric. Ecosyst. Environ. 230, 150–161.
UNDESA, 2022. 4 per 1000: Strategic Plan.
Upson, M.A., Burgess, P.J., 2013. Soil organic carbon and root distribution in a temperate arable agroforestry system. Plant and Soil 373, 43–58.
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.
van Genuchten, M.T., Nielsen, D.R., 1985. On describing and predicting the hydraulic properties of unsaturated soils, pp. 615–628.
Varah, A., Jones, H., Smith, J., Potts, S.G., 2013. Enhanced biodiversity and pollination in UK agroforestry systems. J. Sci. Food Agric. 93, 2073–2075.
Virtanen, P., et al., 2020. SciPy 1.0: fundamental algorithms for scientific computing in python. Nat. Methods 17, 261–272.
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. Chang. Biol. 26, 3759–3770.