How do management interventions influence soil carbon storage and sequestration in UK woodland?

Sally Bavin[1]

[1]Woodland Trust, Kempton Way, Grantham, Lincolnshire, NG31 6LL.

Correspondence: **conservation@woodlandtrust.org.uk**

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1 Foreword

A key aim of this evidence review is to underpin Woodland Trust policy and practice on woodland creation. It is a companion piece to the evidence review: 'How do soil properties influence carbon storage and sequestration in newly established woodland across the UK?' (Bavin, 2021).

2 Executive Summary

The impact on soil carbon storage of a range of management approaches to woodland creation and beyond were considered.

- **Site preparation:** Cultivation leads to soil carbon loss, especially from organo-mineral soils. The impact of the more intense cultivation techniques on organo-mineral soils (>30% soil disturbance) have recently been estimated: lower intensity cultivation techniques are assumed to lead to smaller soil carbon losses from these soils, but insufficient empirical evidence limit confidence in estimations of carbon savings for low intensity site preparation. Drainage of organo-mineral soils can cause a shift from carbon sink to carbon source. However, the associated enhanced tree growth and related carbon storage in humus layer accumulation may compensate for losses due to increased soil carbon input. The carbon loss due to site preparation has not yet been separated from losses induced by the growth of trees on organo-mineral soil; this is an area of active research.
- **Natural regeneration:** The soil carbon impacts of natural regeneration and planting have not been directly compared. Both lead to similarly substantial gains of soil carbon on arable land. The impact of natural regeneration on soil carbon of former grasslands is less clear; there is no evidence from the UK, and global findings are highly variable. Natural regeneration may reduce but does not eliminate the risk of soil carbon loss from grasslands as might be assumed considering no soil preparation is required. To date, the effect of natural regeneration on organo-mineral soil carbon stocks has not been measured but this is an area of active research. Important losses of soil carbon could result from natural regeneration on organo-mineral soils because of soil priming.
- **Direct seeding:** There have been no experiments testing the impact of woodland creation via direct seeding on soil carbon. Direct seeding on arable land is likely to be a carbon sink due

to low levels of pre-afforestation soil carbon. On organo-mineral soils, the same concerns exist as with natural regeneration with regards soil priming. The additional problem of soil cultivation may be avoidable in the uplands, as adequate establishment from direct seeding in the uplands can be achieved by use of herbicides rather than cultivation. However, the effect of herbicides on soil carbon is unknown and requires investigation.

- **Choice of tree species:** Soil carbon stocks over the entire soil profile are similar among conifers and broadleaves, but the distribution of carbon within the soil profile differs. Soil carbon stocks in the surface litter and humus layer (known as 'forest floor') are generally greater under conifers while deeper mineral soil carbon stocks are greater under broadleaves, suggesting broadleaved trees promote higher input and stabilisation of carbon in the mineral soil if of high clay texture. Native broadleaved trees support a rich soil fauna community which helps to promote stabilisation of carbon in the deep mineral soil. Trees associated with arbuscular mycorrhizal fungi tend to accumulate more carbon in the deep soil, suggesting greater input and potential stabilisation. Trees with nitrogen-fixing symbionts promote greater soil carbon storage, especially in the mineral soil, although with caveats.
- **Tree species diversity:** A higher diversity of tree species contribute to greater soil carbon stocks in the mineral soil due to a variety of complimentary rooting patterns. There are also indirect benefits of woodland diversity for carbon storage such as increased resilience to natural disturbance such as pests and disease, which lowers the risk of future carbon loss.
- **Clear-fell harvesting:** Clear-fell harvesting leads to a loss of soil carbon of around 10% of the soil carbon in the entire soil profile with greatest loss (around 30%) from the forest floor layer. Losses from the deeper mineral soil tend to be smaller but are relatively under-sampled. The losses of soil carbon induced by clear-felling occur gradually, with stocks reaching a minimum 30-50 years into the next rotation, before recovering. On organo-mineral soils the risk of soil carbon loss due to soil disturbance during clear-fell harvesting is greater, but total stocks can recover during the second rotation.
- **Continuous cover forestry:** Continuous cover forestry may reduce carbon losses compared to clear-fell harvesting on mineral soils. A single study of continuous cover forestry on organo-mineral soils, suggested benefits compared with clear-felling. Further research into the likely benefits of CCF on soil carbon in organo-mineral soils is needed.
- **Thinning:** The impact of thinning on soil carbon stocks is relatively small, and often insignificant. The intensity of thinning appears to influence the level of impact, but evidence is not strong enough to prescribe threshold thinning intensities to prevent loss of carbon. Broadleaved woodlands are likely to be more susceptible to soil carbon loss from thinning than coniferous woodland due to difference in litter quality. Thinning can increase the stability of individual trees within stands which reduces the risk of carbon loss from soil disturbance due to wind-throw. Long-term field experiments studying soil carbon stocks under different intensities of stand thinning are needed.
- **Removal of harvest residues:** The removal of harvest residues usually has an overall negative effect on forest soil carbon stocks compared with retaining them onsite. Soils with an already very high carbon content such as carbon-saturated clay soils tend to lose more carbon if residues are left on site than if they are removed, possibly due to soil priming. Stump removal is the most negative activity and leads to soil carbon loss in all soil types, particularly organo-mineral soils.
- **Minimum intervention:** By avoiding carbon losses associated with harvesting, unmanaged forests tend to have higher overall onsite carbon stocks than managed forests. Resuming

management will likely impact soil carbon stocks, although this could be negligible with low intensity thinning, avoiding ground disturbance. Little information is available for the implications of minimal intervention.

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3 Introduction

The Committee on Climate Change (CCC) recommend new woodland creation and bringing existing woodland into management as two actions necessary to achieve the UK's commitment to reaching net zero carbon emissions by 2050 (Committee on Climate Change, 2020). If these actions are to deliver a reliably effective method of providing additional carbon sink capacity, it is important to understand how different interventions involved with creating new woodlands and managing existing woodlands affect their carbon balance. As woodland soils contribute a large proportion of carbon storage in woodlands, it is especially important to understand the impacts of interventions on woodland soils. This review assesses the evidence for the impact of management interventions on the soil carbon balance of new and existing woodlands.

The carbon balance of soil is affected by the inputs and losses of carbon within organic matter (the products of photosynthesis). Inputs of organic matter are from litter, deadwood, root turnover, root exudates and microbial and fungal biomass. Soil carbon loss occurs when soil organic matter is decomposed, releasing carbon dioxide. Soil carbon can accumulate when the rate of organic matter input is greater than the rate of its loss. A distinction must be made between the accumulation of soil carbon and the stabilisation of soil carbon. Accumulated soil carbon is vulnerable to being lost to the atmosphere again via decomposition whereas stabilised soil carbon is the portion of accumulated carbon which has formed complexes with soil minerals which reduces this vulnerability. The ideal situation for woodland soils to deliver effective carbon sequestration is a high input rate and low output rate, leading to accumulation, plus a high stabilisation rate leading to effective long term carbon storage. Management activities at the woodland creation stage, and ongoing woodland management can influence inputs, outputs and stabilisation of soil carbon. The focus of this review is on the dynamics of woodland soil carbon and its exchange with the atmosphere as CO_2. Although other greenhouse gases can affect the overall impact on the climate, they are not the focus of this review. Therefore, the term 'net carbon sequestration' when used in this review refers to the balance between CO_2 capture and CO_2 emissions only, rather than CO_2 equivalent. Where other greenhouse gases are being considered this is stated in the text.

4 Methods

A systematic literature search was conducted using Google Scholar. Search terms: "soil" AND "carbon" AND "forest/woodland" AND "name of management intervention". Reference lists were checked to identify additional studies and included where relevant. All sources of relevant evidence were included, such as primary experimental studies, literature reviews, grey literature.

5 Approach to woodland establishment

5.1 Site preparation and cultivation

Intentional soil disturbance (cultivation) is a common form of site preparation used in forestry to improve successful tree establishment. It improves growth and survival of tree seedlings by removing competing vegetation, improving soil temperature, moisture or aeration, increasing nutrient availability, and reducing potential damaging effects by insects (e.g. weevils) and small mammals (Paterson & Mason, 1999). However, the conditions created by cultivation also increases the rate of decomposition of soil organic matter and therefore the loss of soil carbon, especially from the unprotected (labile) soil carbon. The physical process of soil disturbance during cultivation also destabilises soil carbon from clay mineral associations and breaks up soil aggregates – both of which would otherwise hold carbon in the soil by preventing it from being decomposed (Six et al., 2002). The increase in decomposition after cultivation is likely to be very pronounced on dry sandy soils compared with clay rich sites due to the lower ability of sandy soils to stabilise carbon (Carlyle, 1993). Similarly, on organo-mineral soils, the accumulated organic layer is not stabilised but is preserved due to a low rate of decomposition under waterlogged conditions, therefore it is vulnerable to loss when aerated via cultivation.

Site preparation encompasses a variety of soil cultivation practices with different intensities of soil disturbance. Previous reviews have concluded that the more intense the soil disturbance, the greater the loss of soil carbon (Jandl et al. 2007; Mayer et al. 2020). However, the studies cited to support this trend do not provide a clear consistent relationship between intensity of site preparation and soil carbon loss for similar soil types, such that accurate predictions could be made about the outcome of any proposed intervention for a particular site.

Mechanised cultivation disturbs much more soil than hand treatments. Based on the tendency for intense soil disturbance to lead to greater soil carbon loss, it is logical to assume that ploughing is likely to cause the greatest carbon losses, whereas hand screefing is likely to cause the smallest carbon losses (Jandl et al., 2007). This assumption has not been validated in a controlled experiment with all of the listed techniques compared like-for-like. However, a global meta-analysis of 120 sites reveals a significant effect of 'high' vs 'low' disturbance during soil preparation for new woodland creation. On average, 20 years post-planting, soil carbon stocks increased by 19% at low disturbance sites, compared with only 4% at high disturbance sites (Laganière et al., 2010). The smaller increase in soil carbon after high disturbance suggests the increased inputs from the new trees are being offset by increased losses.

5.2 Site preparation and cultivation on organo-mineral soils

Soils with higher carbon contents lose more carbon following site preparation disturbance than sites with low carbon content (Johnson, 1992). On organo-mineral soils, mechanical ground preparation and drainage is often imposed to create the conditions enabling trees to grow. These practices enhance aeration, organic matter oxidation and soil carbon loss (Mayer et al., 2020).

The expected impact of soil cultivation on soil carbon of peaty soils using different cultivation techniques have recently been quantified based on measurements from chronosequence studies (Vanguelova, in preparation). The lower intensity cultivation techniques, such as manual screefing, are assumed to lead to smaller soil carbon losses from these soils, but measurements of soil carbon loss resulting from these techniques are lacking. High intensity cultivation, such as agricultural ploughing, is estimated to lead to 50% loss of soil carbon (Vanguelova, in preparation). The relationship between the degree of soil disturbance and the loss of carbon from peat soils during ground preparation in an active area of research. Until recently, this information was rarely reported in the literature so there is very limited data allowing the comparison of the effect of different ground preparation practices on the soil carbon and GHG balance (Vanguelova et al. 2018). For example (Mojeremane et al., 2012) measured the effects of mounding compared with no mounding on the fluxes of greenhouse gases from the surface of a peaty gley soil, but no alternative cultivation methods were tested for comparison. Soil carbon dioxide efflux was not affected by mounding. Taking into account effects on methane and nitrous oxide, overall, mounding reduced CO_2-equivalent emissions by ~8 per cent in year 1, but had no effect on emissions in year 2.

Short-term impact studies can provide valuable insight into responses of soils to intervention but fail to demonstrate the longevity of any effect (Vanguelova et al. 2018). More research is needed on long-term effects of different types of soil disturbance and their intensity on soil carbon stocks in peatland forests (Mayer et al., 2020). New information from studies distinguishing between disturbance effects from different management techniques is needed to underpin best management practices and guidance for soil carbon storage and greenhouse gas mitigation.

The negative effect of site preparation on soil carbon stocks is expected to be overridden by enhanced tree growth and increased carbon input to soil, such that the long-term effect of site preparation may be reduced (Mayer et al., 2020). UK Forestry Standard (UKFS) indicates managers should "minimise the soil disturbance necessary to secure management objectives, particularly on organic soils" (Forestry Commission, 2017). However, Smith et al. (2007) warned this guidance must be regarded as very preliminary as it is based on limited data and had not been tested against modelled scenario outcomes. This statement still held true in more recent evidence assessments (Vanguelova et al. 2018), but this is an area of active research. Some modelling results are now available which begin to quantify the relationship between intensity of disturbance and amount of soil carbon lost (Creber et al., in preparation.; Vanguelova, in preparation). These show that reducing the level of cultivation disturbance can reduce the time to net carbon benefit on organo-mineral soils, by 3-5 years depending on peat depth. When considering the years to net carbon benefit, medium disturbance cultivation options should be limited to peat depths of <10 cm to maximise the delivery against Net Zero ambitions. There is a need for more empirical data to underpin these modelling scenarios, especially for the minimal disturbance scenarios which are based on assumed rather than measured percentages of soil carbon loss.

Using a 'conservative approach', the Woodland Carbon Code (WCC) also advises to minimise disturbance when planting on organo-mineral soils (West, 2011). The WCC guidance currently prescribes a percentage of initial soil carbon to subtract from the predicted above-ground carbon sequestration of a planting scheme according to pre-planting cultivation method and whether the soil is mineral or organo-mineral (West 2011). The basis for the current estimates of soil carbon loss percentages is unclear. However, the WCC guidance will soon be updated based on new modelling (Vanguelova, in preparation). Recent findings from studies of low-disturbance native tree planting on organo-mineral soils (Friggens et al., 2020; Warner et al., 2021) seriously challenge the estimate of zero loss of carbon from organo-mineral soils under minimal soil disturbance. This work suggests the WCC guidance may not be as conservative as it once appeared. A more conservative approach would be not to credit carbon sequestration benefits of woodland creation on organo-mineral soils unless they are proven by soil monitoring.

Both site preparation and tree growth affect soils. It is therefore difficult to determine to what extent each is causing loss of soil carbon in each scenario. Chronosequence studies use sites with similar ecological characteristics but different ages to substitute space for time. It is an approach that has been used to assess net soil carbon flux and studies have tended to assume that the soil carbon lost over the decades following afforestation of organo-mineral soils was all the result of site preparation (Vanguelova et al., 2019). However, there are several ways in which the growth of trees can affect soil carbon, such as drying the soil by transpiration, and soil priming (Dijkstra & Cheng, 2007; Mitchell et al., 2007). Soil priming is a biochemical process whereby the input of new soil carbon stimulates decomposition of old soil carbon (Liu et al., 2020). The effects need to be disentangled in future research so that the benefit of minimizing the intensity of site preparation is not overstated.

5.2.1 Drainage of organo-mineral soils

Waterlogged conditions can prevent or impede tree growth, so drainage is a common practice to reduce the water table under organo-mineral soils prior to planting. The UKFS guidance advises to "consider the potential impacts of soil disturbance when planning operations involving cultivation, harvesting, drainage and road construction" (Forestry Commission, 2017). Drainage significantly increased CO_2-equivalent emissions by ~18−29% in a 2 year study of organo-mineral soils in northern England (Mojeremane et al., 2012). The authors consider it likely this pattern of loss would have continued beyond the 2 years (Mojeremane et al., 2012). Losses of carbon also occur as dissolved organic carbon (DOC) which washes into rivers and the ocean, a large portion eventually becoming atmospheric CO_2. Drainage of high-carbon soils for upland conifer plantation forestry leads to greater loss of DOC than previously thought and is the main human activity influencing DOC exports from British river catchments (Williamson et al., 2021). This important pathway may have been underestimated in forestry carbon models and should be incorporated in future assessments of the carbon sequestration potential of forest planting.

Drainage of high carbon content soils is one of the largest potential causes of carbon loss and may cause a shift from a carbon sink to a carbon source (Morison et al., 2012). However, because drainage allows tree growth on areas which were otherwise unsuitable for trees, the carbon loss from drainage is at least partly compensated by the higher inflow of carbon into the system through increases in plant biomass and primary production, but this appears highly variable. In their review, (Jandl et al., 2007) summarise that both gains and losses in overall carbon balance of afforested peatlands have been reported. This is to be expected as so many site-dependent factors will affect the balance, such as depth of peat, growth rate of trees etc. For woodland where the primary objectives are biodiversity recovery and carbon sequestration, drainage is unlikely to be used.

5.3 Natural regeneration

Natural regeneration compared with planting as a method of woodland establishment has received no attention in previous reviews on management of woodland soil carbon (Jandl et al., 2007; Laganière et al., 2010; Mayer et al., 2020). There are no published studies directly comparing the methods, however this is an area of active research; evidence addressing this gap should become available in 2022 (Vanguelova, personal communication).

As demonstrated at Rothamsted (UK), natural regeneration of native woodland on arable land has the potential to double or even treble soil carbon stock compared to baseline measurements in a little over a century, with very high rate of initial accumulation, and substantial accumulation continuing into the second century (Poulton et al., 2018). The rate of increase is similar to that recorded for planted native woodland on arable land (Ashwood et al., 2019) suggesting natural regeneration does not have a major impact on soil carbon accumulation, at least on lowland ex-arable sites. On such sites, avoiding soil disturbance is not a concern because the soil is already disturbed.

The impact of natural regeneration on grassland soil carbon is less clear; there appears to be no evidence from the UK. A meta-analysis of 142 studies of shrub encroachment on grasslands worldwide revealed huge variability in the effect on soil carbon. Changes ranged from −50% to + 300%. Soil carbon increased in semi-arid and humid regions and showed a greater rate of increase under leguminous encroaching shrubs. Soil carbon decreased in silty and clay soils but increased in sandy soils. Gains were correlated positively with annual rainfall on a scale of 200 to 1000mm (Li et al., 2016), yet opposite results have been reported on similar scales of precipitation in the US, (Jackson et al., 2002) and South Africa (Mureva et al., 2018).

Natural regeneration of Norway spruce forest on abandoned Alpine meadows (previously mown once or twice a year) lead to a redistribution of carbon in the soil profile based on a chronosequence of successional plots, from meadow to high forest (Thuille & Schulze, 2006). There was increased carbon stored in the newly formed litter and humus layers (not previously present in the meadows) but a decline in mineral soil carbon stocks of 20–40% during forest development. The location of the carbon in the soil profile is important for long term carbon storage, as carbon in the mineral soil is more stable over longer periods than in organic matter accumulated at the surface. Even after 100 years of forest development, only one of the successional forests had reached more than 90% of the mineral soil carbon stock in the meadow sites. Likely mechanisms for the loss of carbon from the mineral soil were identified: 1. Decomposition of the fine root system of the former meadow. 2. Changes in mycorrhiza as the vesicular-arbuscular mycorrhizae of grasslands, are replaced by ectomycorrhizae and ericoid mycorrhiza, 3. Reduction in soil fauna activity in the acid soils developing under spruce forests, thus reducing the mixing of organic material into the mineral soil. The situation might be different with some deciduous tree species with litter which is more readily integrated into the mineral soil by earthworms, potentially increasing mineral soil carbon stocks (Thuille & Schulze, 2006).

The wide range of results from across the world show the importance of the local context, underscoring the need for studies in the UK. However, natural regeneration as a method of woodland creation on former grasslands does not eliminate the risk of soil carbon loss, despite no soil preparation being required.

5.3.1 Natural regeneration on organo-mineral soils

The heavy focus in the literature on site preparation as the cause of soil carbon loss on organo-mineral soils leads to the suggestion that natural regeneration may have less impact on soil carbon stocks than more disruptive planting methods as no soil disturbance is involved. For example, natural regeneration of the British uplands has been identified as an economically viable climate mitigation strategy (O'Neill et al., 2020). This analysis compared a planting scenario with a natural regeneration scenario. The planting scenario took into account soil carbon loss from ground cultivation, but held the implicit assumption that the natural regeneration scenario will lead to no loss of carbon from the soil, potentially drastically over-estimating the effectiveness of the strategy. See also (Fletcher et al., 2021).

The effect of natural regeneration on organo-mineral soil carbon stocks has not been measured in any empirical studies (Berdeni et al., 2020). However, modelling of native broadleaf natural regeneration scenarios on peatland suggests carbon benefits with or without low intensity ground preparation (Vanguelova et al., 2021). The benefits are predicted to be greater if no ground preparation is required, but the model assumes no loss of soil carbon with no cultivation. It is very likely that the absence of soil disturbance from site preparation could *reduce* the carbon losses relative to intensive site preparation, but the magnitude of this potential reduction is unknown. It is unlikely to reduce the soil carbon loss to zero. Important losses could still take place because of top-down ecological effects such a soil priming (Trivedi et al., 2018) as trees act as ecosystem engineers (Mitchell et al., 2007) with interactions between soil and tree roots substantially accelerating soil carbon decomposition (Dijkstra & Cheng, 2007). Losses of soil carbon substantial enough to cancel out sequestration in trees over 39 years (Friggens et al., 2020) and 20 years (Warner et al., 2021) have been observed as a result of hand-planting on organo-mineral soil in Scotland. The minimal soil disturbance involved suggests other factors are triggering soil carbon loss, and therefore that natural regeneration could produce similar effects.

Perks and Vanguelova (2020) identify "assessing, documenting and comparing the carbon sequestration dynamics from assisted natural regeneration, compared with planting native woodland" as a current research need. They highlight the importance of the recent initiative *Future Woodlands Scotland* https://www.futurewoodlands.org.uk/" for beginning to provide "valuable information". A new UK research project has recently been initiated with plans to study the above and belowground carbon dynamics of natural regeneration across 60 sites including a range of soil types (Vanguelova, *personal communication).*

Wet woodland habitats can form naturally on peat in valley bottoms with impeded drainage. The dominant trees will be those tolerant of waterlogging: alder and willow species. Wet woodland has been lost from our landscapes on a large scale. Restoring this habitat in appropriate places could provide a double benefit for carbon sequestration and storage in that peat can continue to form in the waterlogged conditions as well as supporting tree growth. However, more research is needed on the likely GHG balance for the habitat as there is uncertainty around the potential methane and nitrous oxide emissions (Gregg et al., 2021).

5.4 Direct seeding

Direct seeding is a potential method for encouraging tree growth in areas devoid of adequate seed source. There have been no experiments testing the impact of woodland creation via direct seeding on soil carbon. On arable land, the soil carbon impacts are likely to be similar to planting and natural regeneration of native broadleaf woodland – a strong carbon sink.

For direct seeding on organo-mineral soils, the same concerns about soil priming exist as with natural regeneration. If soil cultivation is an unavoidable part of the direct seeding process, this increases the likelihood of large soil carbon losses with this method due to the soil disturbance. Soil cultivation prior to sowing has been shown to significantly increase the density of saplings in the first four years (Willoughby et al., 2019). Nevertheless, even without cultivation, a median sapling density of 10,100 saplings per hectare was achieved after 7 years, compared to 12,300 saplings per hectare with the cultivated plots. It is important to note that in this experiment, the uncultivated plots had been treated with herbicide prior to sowing. It is possible that herbicide application also leads to impacts on soil carbon.

Further research is needed to measure the impact on soil carbon of upland woodland created by direct seeding. It would be useful to build on the work of (Willoughby et al., 2019) by investigating whether sowing onto untreated upland vegetation can be effective, or if herbicide is necessary for successful establishment. If so, the effects of herbicide on soil carbon stocks need to be investigated.

5.5 Choice of tree species

There are consistent tree species effects on soil carbon stocks. The effects are clearest for forest floor carbon stocks, whereas impacts on the deeper mineral layers are less consistently reported (Vesterdal et al., 2013). It is the distribution of carbon between forest floor and mineral soil rather than total carbon stock which tends to differ according to tree species. This suggests that some species may be better for sequestration of carbon in stable form in the mineral soil.

Differences in total soil carbon stocks over the entire soil profile are often insignificant among conifers and broadleaves (Mayer et al., 2020). However, differences have been reported in the rate of accumulation, stability and distribution of soil carbon. Soil carbon stocks in the surface litter and humus layer (known as ‘forest floor’) are generally greater under conifers while mineral soil carbon stocks are greater under broadleaves (Vesterdal et al., 2013). The stability of soil carbon also appears to be greater under broadleaved tree species. This is partly due to greater carbon incorporation and stabilisation in mineral soil related to the more abundant soil macrofauna communities under these tree species (Mayer et al., 2020). The influence of root turnover in the mineral soil remains to be addressed in detail using well-designed experimental platform (Mayer et al., 2020).

Meta-analyses have addressed the question of which types of tree lead to greatest carbon sequestration in the soil, but they are generally global in scope and the species or groupings included can be unhelpful for applying results to the UK context (Laganière et al., 2010; Li et al., 2012). A meta-analysis of 90 studies of monoculture plantations from around the world found the most accurate way of grouping trees when it comes to soil sequestration rates is as ‘deciduous broadleaf’, ‘evergreen broadleaf’ and ‘evergreen conifer’. The soil sequestration rates between the deciduous broadleaf and evergreen conifer groups are the most relevant for the UK and were not significantly different from each other (Hou et al., 2020).

For effective long-term sequestration, the aim is to increase the stable soil carbon pool. The stability of carbon increases with depth (Lorenz & Lal, 2005). Therefore, the aim is to select tree species which will provide high inputs of carbon to the deep soil – species with deeper and thicker root systems that are high in chemical recalcitrant compounds like suberin (Lorenz & Lal, 2005). A high surface input of organic matter (litter) is also important, but to contribute to the stable subsoil the carbon needs to be transported to deeper soil layers. The activity of soil fauna, particularly earthworms is important. The influence of earthworms on soil carbon stocks is not to increase the overall stock, but to increase the portion which is stabilised (e.g., in earthworm casts) (Thomas et al., 2020). Broadleaved woodlands have been shown to support greater abundance, diversity and functional complexity of earthworm communities and other soil invertebrates compared to coniferous woodlands (Barsoum & Henderson, 2016) and compared to arable and pasture land (Ashwood et al., 2019).

An investigation of potential ecological 'replacement' trees in the UK as a management response to the impact of disease on native ash *F. excelsior* populations, showed soil carbon in the top 10cm was significantly lower under sycamore *A. pseudoplatanus* than under ash and oak *Q. petraea* (Mitchell et al., 2020). However, the mechanism was not clear. Root-derived carbon inputs are important contributors to soil carbon stock, as they can be equal to or greater than leaf litter inputs (Keller et al., 2021). The roots of different tree species form interactions with different types of soil fungi, mainly arbuscular mycorrhizal (AM), and ectomycorrhizal (ECM). Variation in root-derived soil carbon accumulation due to tree mycorrhizal association may be a key control of soil carbon stabilisation in forests. AM association appears to promote greater root-derived carbon in the stable mineral-associated pools in the subsoil, compared to ECM association, which promotes higher carbon stocks in the topsoil (Craig et al., 2018; Keller et al., 2021). The mechanism is an accumulation of microbial residues at depth in AM-dominated soils (Craig et al., 2018). Because of this pattern of distribution, previous work sampling only at shallow depths had concluded that ECM dominated plots contained greater soil carbon (Averill et al., 2014).

ECM associated trees tend to dominate on soils which are high in carbon and low in nitrogen, typical traits of British upland soils. It is the low nitrogen levels which are the important factor conferring advantage to ECM associated trees, rather than the high carbon content (Zhu et al., 2018). A list of UK native tree species and their fungal associations is provided in appendix 1.

5.5.1 Diversity of tree species

Some studies have suggested a positive effect of tree diversity on soil carbon stocks. Species diversity is a weaker driver than species identity, however diversity increases soil carbon stocks in the 20-40 cm layer of mineral soil, possibly due to belowground niche complementarity driving higher root carbon input (Dawud et al., 2016). In Swedish forestry, topsoil carbon storage was 11% greater in stands with five than with one tree species (Gamfeldt et al., 2013). It may be advantageous to identify the specific mixtures of tree species or functional groups of tree species that are most conducive to sequestration of soil C. More experiments comparing plots with different tree species diversity are needed (Mayer et al., 2020). Further research is being conducted at the BangorDIVERSE (UK) tree diversity experimental plot at Bangor University.

Indirect benefits of woodland diversity for carbon storage are more universally agreed, such as increased resilience to natural disturbance such as pests and disease, which lowers the risk of future carbon loss (Jandl et al., 2007; Mayer et al., 2020).

5.5.2 Nitrogen fixing tree species

It is well documented that trees with N-fixing root associates consistently lead to increases in soil carbon stocks, suggesting including N-fixing species in woodland creation could be an important contribution to sequestering carbon in soil (Mayer et al., 2020). For example, significant increases in soil carbon storage (+12%) occurred in response to nitrogen fixing vegetation, as revealed in a meta-analysis of 72 northern temperate forests (Nave et al., 2009). This increase was in the mineral soil rather than the forest floor suggesting the effect is important for stable carbon storage. In the UK, the selection of nitrogen fixing tree species is limited to alder *Alnus glutinosa* and some shrubs such as gorse *Ulex europaeus.*

Another benefit of nitrogen fixing species may be the possibility for increased growth (and therefore inputs of organic matter to the soil) on nitrogen poor soils, where other trees might struggle to grow (Mayer et al. 2020). However, widespread planting of nitrogen fixing species could lead to increased nitrous oxide (N_2O) emissions from soil, which has the potential to counter-act the benefit of increased carbon sequestration. The predicted net CO_2 equivalent greenhouse gas balance for planting nitrogen fixing species is uncertain but theoretical modelling suggests this is an important area for further study (Kou-Giesbrecht & Menge, 2019).

6 Approach to ongoing woodland management

6.1 Clear-fell harvesting

Clear-fell harvesting involves removing all trees from a stand and removing the timber. This causes several changes to the soil conditions which can affect soil carbon stocks. The removal of the trees means the continuity of litter input is interrupted. The soil is no longer shaded by the tree canopy, which can increase soil temperature and microbial respiration. In addition, machinery causes soil disturbance (Jandl et al., 2007). These conditions generally lead to a loss of around 10% of the soil carbon in the entire soil profile with greatest loss from the forest floor as opposed to the mineral soil (Mayer et al., 2020). However, mineral soil losses have been reported, and the deeper soil layers are under-sampled. Sampling deep soil represents one of the best opportunities to reduce uncertainty in the understanding of the response of soil carbon to forest harvest (James & Harrison, 2016).

Meta-analyses estimate clear-fell harvesting significantly reduces soil carbon by an average of between 8% (Nave et al., 2010) and 11.2% (James and Harrison, 2016). Forest floor carbon stocks decline consistently by around 30% (Nave et al., 2010; James and Harrison, 2016). Mineral top-soils (0-15 cm depth) show small but significant carbon losses of 3.3% (James and Harrison, 2016). A significant loss of 17.7% of soil carbon is estimated in very deep soil (60–100+ cm) after harvest. However, only 21 of the 945 total studies examined this depth; study methods were highly skewed toward surface sampling, with a maximum sampling depth of 36 cm, on average (James and Harrison, 2016).

The process of losing soil carbon due to harvesting is not an instantaneous event. The change in conditions leading to net carbon release can persist for decades after a harvested stand has been restocked, until the balance of inputs exceeds the outputs again and the carbon stock can begin to recover. It took 32 years for carbon stocks to reach their minimum after harvest and restocking in a US chronosequence study (Diochon et al., 2009). By which point stores were approximately 50% of an intact old-growth forest reference plot. After approximately 100 years, carbon storage approached the range of the intact forest. Similarly, Prest et al., (2014) found soil carbon losses from the mineral soil are evident at least 35 years following harvesting, compared to 110 year old reference stands. The lowest soil carbon stocks were in 43-year-old stands in a chronosequence in

Scotland, also supporting a timescale of several decades of loss before soil carbon stocks begin to recover (Ražauskaitė et al., 2020). Time to recovery varies with soil type. Some stands take at least 75 years, but the majority recover within 50 years of harvest and restock (James & Harrison, 2016).

The level of visible soil disturbance from clear-felling operations correlated with the severity of soil carbon depletion from the mineral soil at a depth of 10–30cm in a small-scale study (Zummo & Friedland, 2011). Plots which showed visible signs of severe soil disturbance contained 25% less total carbon than the least disturbed plots. All plots were felled in the same way, yet visible effects on the soil were sufficiently different to be separated into a high to low gradient. This suggests the actions of individual forestry workers or local soil conditions have an important impact on soil carbon loss during harvesting. There is a good practice technical guidance note produced by Forest Research on protecting soil during harvest (Murgatroyd & Saunders, 2005).

On organo-mineral soils, the risk of soil carbon loss due to soil disturbance during clear-fell harvesting is greater. Clear-felling reduces the carbon stock in organo-mineral soils but this loss can be compensated in the litter layer by 40 years into the second forestry rotation (Vanguelova et al. 2018). Gas flux data showed that clear-felling a first rotation stand of conifers on peaty-gley soil resulted in an average annual loss of 13.38 tC ha^{-1} over the first 4 years, while a mature, unharvested control stand continued to take up 29.38 tC ha each year on average. The largest carbon loss was during the first year after clear-fell. At the end of the 4-year study, the clear-felled stand remained a weak source of CO_2 emissions, but was predicted to become a sink the following year (5 years after clear-felling) (Xenakis et al., 2021). This study illustrates the balance between respiration and photosynthesis for the stand during the first few years after harvest and restock. It does not show that the soil stops losing carbon after 5 years, but that this is when annual aboveground growth (in ground vegetation and replanted trees) becomes substantial enough to counteract annual losses, so that total ecosystem carbon stock can begin to reaccumulate. It does not show that the total carbon emissions since clear-fell (120 tC ha^{-1}) have been recaptured by this point.

6.2 Continuous Cover Forestry

Continuous cover forestry (CCF) is a method of harvesting which involves selectively removing a proportion of mature trees, leaving others standing and allowing natural regeneration or restocking, leading to a stand of mixed ages. CCF reduces the opportunity for soil disturbance during site preparation, but it usually takes several years, even decades, for a new tree canopy to develop. During this time, carbon losses by decomposition can be higher than litterfall inputs, causing a net loss of soil carbon (Blanco, 2018).

Gradual continuous cover restoration maximises on-site carbon stocks during the restoration of plantation on Ancient Woodland sites (PAWS), compared to clear-felling the plantation trees (Colls, 2006). The carbon consequences of four different restoration management scenarios for PAWS were estimated using the C-Flow carbon accounting model.

The long-term soil carbon impacts of CCF are difficult to assess and are yet to be empirically measured (Jandl et al., 2007; Morison et al., 2012). There is no clear consensus when comparing CCF with clear-fell harvesting (Clarke et al., 2015; Mayer et al., 2020). Long term forest monitoring plots in France showed uneven-aged stands sequestered more soil carbon over a 15 year period than the even-aged stand (Jonard et al., 2017). Most studies report short term changes in soil carbon over less than 15 years, whereas changes can take place over multiple decades. Again, the issue of shallow sampling leads to greater uncertainty on the effects of harvesting techniques on carbon in the deep soil (Mayer et al., 2020).

6.2.1 Continuous cover forestry on organo-mineral soils

Findings from Clocaenog Forest, North Wales shows CCF could increase the stable carbon in the mineral layers under organo-mineral soils (Vanguelova et al. 2018), and in the organic horizon (Lanfranchi, 2020) compared with conventional clear-fell forest management. Further research at a range of sites would be beneficial to improve confidence in these findings. Studies of CCF on peaty soils are scarce; CCF is rarely practiced on peaty soils, as restructuring after canopy closure exposes a stand to increase wind risk and rooting in peaty soil is relatively poor and trees less stable (Vanguelova et al. 2018).

6.3 Thinning

Thinning is the reduction of stem density of a stand by removing a proportion of stems to increase radial growth of remaining trees. Thinning causes a temporary change in microclimate at the soil surface affecting the rate of decomposition. At colder sites, the increased levels of sunlight can warm the soil, stimulating decomposition, whereas at hotter sites, the removal of shade leads to drying of the soil and decreased decomposition. Thinning also causes a temporary reduction in litterfall, but the brash left onsite after thinning operations may compensate for this. These factors explain why the effects of thinning on soil carbon stocks are not consistent among published studies and are often not significant (Blanco, 2018; Mayer et al. 2020).

Meta-analyses have found no overall significant thinning effect on soil carbon stocks (Zhang et al., 2018; Zhou et al., 2013). In the initial 2 years after thinning, soil respiration significantly increased but this decreased back to the original level gradually (Zhang et al., 2018). The intensity of thinning affects the impact on soil carbon. Light thinning (≤33% removal) increased soil carbon by 17% whereas heavy thinning (≥65% removal) decreased soil carbon by 8%; moderate thinning (33–65% removal) did not alter soil carbon stocks (Zhang et al., 2018). Broadleaved woodland is more vulnerable than coniferous woodland to increased soil carbon respiration as a result of thinning due to difference in litter quality (Zhang et al., 2018). It is not yet possible to draw conclusions in terms of threshold thinning intensities to prevent loss of carbon. Long-term field experiments to study soil carbon stocks under different intensities of stand thinning are needed to achieve that level of understanding (Zhang et al., 2018). Overall, the impact of thinning on soil carbon stocks appears small (Mayer et al., 2020) and the positive effect of thinning is to increase the stability of individual trees within stands which reduces the risk of disturbance from wind-throw (Jandl et al., 2007).

6.3.1 Thinning on organo-mineral soils

There is a paucity of recent published literature on impacts of thinning operations on soil carbon stocks and GHG balance of peaty soils (Vanguelova et al., 2018). Morison et al. (2010) reported no impacts of thinning, however this needs to be confirmed for UK conditions. There is little further information on the impacts of thinning on peaty soils, possibly because the bulk of the UK forest stock is well past the time of thinning. It may also be because thinning on peat soil is not always carried out as it leads to excessive windthrow on shallow rooting forests on peat (Vanguelova et al., 2018).

6.4 Removal of harvest residues

Forest harvest residues are the leaves/needles, branches, twigs, low-quality or small- diameter stems, bark, dead wood, stumps and roots that are usually left on the forest floor after harvesting because they are of low commercial value. The increasing demand for renewable energy sources has increased interest in the removal of harvest residues for use as biofuel. This practice is known as whole tree harvesting. The potential carbon emissions saved by replacing fossil fuels with bioenergy are beyond the scope of this review. This review will consider just the impact of removing harvest residues on the carbon stocks of the woodland soil.

Several global meta-analyses have investigated the effects of whole tree harvesting and produced divergent findings. (Hume et al., 2018) found no difference in the impact of whole tree harvesting compared with stem only harvesting; neither had any significant impact on forest floor or mineral soil carbon stocks. (Achat et al., 2015) found stem only harvests caused no overall change in carbon stock over the soil profile; a decrease in carbon stock in the forest floor was compensated by an accumulation of carbon in mineral soil layers. In contrast, whole tree harvesting led to carbon losses in all soil layers. Similarly, Wan et al., (2018) found retention of harvest residues leads to greater (+8.2%) soil carbon storage in 0–20 cm mineral soils, compared to when residues are removed. The trend was consistent across broadleaf and coniferous forests. Seven years after clearfelling, woodchipped harvest residues improved transfer of carbon to the mineral soil compared to traditional brash retention at a PAWS site on a clay soil (Pitman & Peace, 2021).

Wan et al., (2018) showed that soil properties (soil clay content and carbon concentrations) affected the soil carbon response to residue retention. Against the general trend, the retention of harvest residues appeared to reduce mineral soil carbon stocks in some high clay content soils. (Wan et al., 2018) suggest this may be due to increased decomposition of existing soil carbon stocks via a priming mechanism. This suggests soil texture and soil carbon concentrations should be considered when deciding whether to remove or retain harvest residues.

Further long-term experiments are necessary assess the long-term impact of intensified forest harvesting on soil carbon stocks (Clarke et al., 2015). However, Mayer et al. (2020) summarise: the removal of harvest residues has an overall negative effect on forest soil carbon stocks. Soil carbon losses are largest after whole tree and stump harvesting operations. On nutrient-poor sites, this might have severe consequences for fertility, productivity and long-term carbon sequestration by limiting tree growth and therefore future carbon inputs to the soil from litter and roots.

6.4.1 Removal of harvest residues on organo-mineral soils

Contrary to the general trend observed for mineral soils, there is extensive evidence that the retention of forest residues on sites with organo-mineral soils can lead to losses of soil carbon by promoting decomposition of the peat layer (Vanguelova et al., 2018). For example, on a peaty gley soil at Kielder Forest, northern England, plots subjected to whole tree harvesting of the first rotation of Sitka spruce 28 years earlier, had higher soil carbon content than equivalent plots where stem only harvesting had taken place and brash residue retained (Vanguelova et al., 2010). Therefore, removing harvest residues (brash) on peaty soils is a positive management decision in terms of preserving soil carbon. However, stump removal results in high loss of carbon from both shallow and deep peat soils due to the soil disturbance involved with removing the stumps (Vanguelova et al., 2018). Experimental evidence from upland Wales showed that stump harvesting caused almost three times higher loss of soil carbon in peaty gley soils compared to mineral sandy brown soils (Vanguelova et al. 2017).

Overall, the effect of brash removal appears positive for soil carbon stocks on organo-mineral soils and negative on mineral soils, whereas stump removal is negative for soil carbon stocks on both organo-mineral and mineral soils, but most strongly negative on organo-mineral soils.

6.5 Minimum intervention

The evidence presented within the harvesting and thinning sections above suggests avoiding harvesting facilitates the accumulation of soil carbon and reduces loss to the atmosphere.

International evidence shows unharvested, old-growth forests can continue to sequester carbon (Luyssaert et al., 2008; Zhou et al., 2006) as well as store it in large amounts over long periods. Measured soil carbon stocks indicate a significant benefit of undisturbed ancient stands in Scotland for soil carbon storage and long-term carbon sequestration (Ražauskaitė et al., 2020) as the ancient forest had significantly more soil carbon than the young forest. Similarly, at an unmanaged ancient woodland in England, old-growth stands had greater soil carbon stocks (11.03 kg C m^{-2}) than young-growth stands which were clear-felled in the 1940s (7.43 kg C m^{-2}) (Hale, 2015). Unmanaged secondary native woodland in the UK attained soil carbon stocks comparable to that in ancient woodland within 50-100 years (Ashwood et al., 2019).

Analysis of global databases (Noormets et al., 2014) revealed that managed forest soils are more likely to be losing soil carbon than unmanaged forest soils. Relatively more of the assimilated carbon is allocated to aboveground pools in managed than in unmanaged forests, whereas allocation to fine roots and rhizosymbionts is greater in unmanaged forests. Therefore Noormets et al. (2014) suggest managing forests for productivity or carbon sequestration requires different approaches. This agrees with the findings from the UK: low intensity management provides high levels of ecosystem services including on site carbon storage according to a synthesis based on 5 forest management approaches ranging in intensity from clear-fell harvesting to non-intervention natural reserves (Sing et al., 2018).

However, neutral results have also been found. Soil carbon stocks in a long-term non-intervention beech forest in Denmark were similar to those in managed beech-dominated forests in the same region, suggesting little potential for soil carbon sequestration from setting forests aside as unmanaged (Nord-Larsen et al., 2019).

There is interest in bringing UK broadleaf woodlands that are currently subject to minimal management back into active management to provide carbon benefits by increasing the annual carbon draw-down rate while providing a source of timber and fuelwood to substitute for fossil fuels and high-carbon construction materials. Resuming harvesting in unmanaged woodland would result in an initial decrease in carbon stocks (Crane, 2020).

7 Conclusion

Soil carbon is an important carbon pool in woodland ecosystems and changes in soil carbon stock can have a substantial impact on the overall carbon balance of a woodland. The quantity of carbon stored in woodland soils can be influenced by woodland management activities, both during woodland creation and during ongoing woodland management.

Activities that tend to lead to losses of carbon from soils include clear-felling and soil disturbance during cultivation. The level of impact from these management practices is related to the type of soil at a site. Organo-mineral soils hold large stocks of unstable carbon at the surface, which makes them particularly vulnerable to losing substantial quantities of carbon as a result of management. Drainage has a negative impact on the carbon stocks of organo-mineral soils soil.

Positive management practices for soil carbon sequestration should aim to achieve three key objectives: minimise losses of the existing soil carbon; promote the accumulation of new soil carbon; and promote the stabilisation of soil carbon to ensure long-term carbon storage.

Planting without soil cultivation and encouraging natural regeneration may reduce soil carbon losses during woodland creation. This could be particularly important for woodland creation on organo-mineral soils, but the size of the reduction is not clear. The benefit of minimising disturbance has not been separated from other mechanisms of soil carbon loss, so it is not clear where the limit of the manager's influence lies. Avoiding clear-felling in favour of continuous cover forestry or minimum intervention management is a positive choice for protecting existing soil carbon and maintaining a continuous accumulation of carbon in the soil. Generally, it is beneficial to retain harvest residues on site to allow carbon to accumulate. The exception is on organo-mineral soils, where the decomposition of brash stimulates decomposition of the peat. To encourage stabilisation of soil carbon, the choice of native broadleaved tree species is beneficial. Their deep roots can input carbon to the deeper layers of the soil, and their litter supports rich communities of soil invertebrates which contribute to stabilising soil carbon.

7.1 Key primary evidence gaps and recommendations for future research

This evidence review identified several key primary knowledge gaps. The majority of evidence on soil carbon and woodland management comes from studies of commercial forestry. More studies focussing on native woodlands in the UK would improve the applicability of the evidence for guiding the UK conservation sector. On organo-mineral soils, the relationship between the intensity of soil disturbance and long term-soil carbon loss needs to be more accurately quantified. Models predicting the relationship require validation with data from field experiments directly comparing various ground preparation techniques on a gradient of disturbance intensity, including low and zero disturbance, natural regeneration and control plots where no woodland is established. Efforts should be made to disentangle the relative contribution to total soil carbon losses from soil priming after tree establishment compared with disturbance from ground preparation. Closing this knowledge gap is necessary to ensure that the benefit of minimising soil disturbance during site preparation is not overstated. This is particularly important in the context of major woodland expansion via natural regeneration in the uplands, and research should aim to quantify likely time-lags before net positive carbon sequestration.

Further research is needed on the feasibility of direct sowing in upland settings without any soil preparation. This would help to clarify the potential of this method as a low-disturbance option for locations where seed source for natural regeneration is limited.

Nitrogen fixing tree species promote carbon sequestration in soils. However, the potential side-effect of increased nitrous oxide emissions from soils remains uncertain and potentially significant for the overall greenhouse gas balance of the soil. Research focussing on our native nitrogen fixing tree species, alder, should aim to quantify this trade-off.

Continuous-cover forestry is likely to reduce soil carbon losses when compared to clear-fell harvesting, but there is no consensus for this effect in the literature. Longer-term studies would reduce this uncertainty. Field experiments studying the impact of different intensities of stand thinning on soil carbon are needed to define threshold thinning intensities to prevent soil carbon loss. In all future research on soil carbon, sampling soil to a greater depth represents one of the best opportunities to reduce uncertainty in understanding of the response of soil carbon to any management intervention.

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9 Appendix

Mycorrhizal types on UK tree and shrub genera, from Brundrett, and Tedersoo (2020).

Arbuscular mycorrhizas (AM)	**Ectomycorrhizas (ECM)**	**Both AM and ECM**
Acer (maples) Cornus (dogwood) Crataegus (hawthorn) Fraxinus (ash) Ilex (holly) Juniperus (juniper) Ligustrum (privet) Malus (apple) Platanus (plane) Pyrus (pear) Rhamnus (buckthorn) Sorbus (rowan, service, whitebeam) Taxus (yew) Ulmus (elm)	Betula (birch) Carpinus (hornbeam) Fagus (beech) Pinus (pine) Quercus (oak) Tilia (lime)	Alnus (alder) Corylus (hazel) Populus (aspen, poplar) Salix (willow)