Qualitative Impact Assessment of Land Management Interventions on Ecosystem Services ("QEIA")

Report-3 Theme-6: Carbon Sequestration

30-June-2023

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This report is one of a set of reviews by theme:

Braban, C.F., Nemitz, E., Drewer, J. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA").* Report-3 Theme-1: Air Quality (Defra ECM_62324/UKCEH 08044)

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Newell Price, J.P., Williams, A.P., Bentley L. & Williams, J.R. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3 Theme-3: Soils (Defra ECM_62324/UKCEH 08044)

Williams, J.R., Newell Price, J.P., Williams, A.P., Bowes, M.J., Hutchins, M.G. & Qu, Y. et al. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3, Theme-4: Water (Defra ECM_62324/UKCEH 08044)

Staley, J.T., Botham, M.S., Broughton, R.K., Carvell, C., Pywell, R.F., Wagner, M. & Woodcock, B.A. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3 Theme-5A: Biodiversity - Cropland (Defra ECM_62324/UKCEH 08044)

Keenleyside, C.B. & Costa Domingo, G. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3 Theme-5B: Biodiversity - Grassland (Defra ECM_62324/UKCEH 08044)

Maskell, L. & Norton, L. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3 Theme-5C: Biodiversity - Semi-Natural Habitats (Defra ECM_62324/UKCEH 08044)

Siriwardena, G.M. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3 Theme-5D: Biodiversity - Integrated System-Based Actions (Defra ECM_62324/UKCEH 08044)

Bentley, L., Feeney, C., Matthews, R., Evans, C.D., Garbutt, R.A., Thomson, A. & Emmett, B.A. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3 Theme-6: Carbon Sequestration (Defra ECM_62324/UKCEH 08044)

Short, C., Dwyer, J., Fletcher, D., Gaskell P., Goodenough, A., Urquhart, J., McGowan, A.J., Jones, L. & Emmett, B.A. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3.7: Cultural Services (Defra ECM_62324/UKCEH 08044)

A list of all references used in the reports is also available as a separate database.

Foreword

The focus of this project was to provide a rapid qualitative assessment of land management interventions on Ecosystem Services (ES) proposed for inclusion in Environmental Land Management (ELM) schemes. This involved a review of the current evidence base by ten expert teams drawn from the independent research community in a consistent series of ten Evidence Reviews. These reviews were undertaken rapidly at Defra's request and together captured more than 2000 individual sources of evidence. These reviews were then used to inform an Integrated Assessment (IA) to provide a more accessible summary of these evidence reviews with a focus on capturing the actions with the greatest potential magnitude of change for the intended ES and their potential co-benefits and trade-offs across the Ecosystem Services and Ecosystem Services Indicators.

The final IA table captured scores for 741 actions across 8 Themes, 33 ES and 53 ES-indicators. This produced a total possible matrix of 39,273 scores. It should be noted that this piece of work is just one element of the wider underpinning work Defra has commissioned to support the development of the ELM schemes. The project was carried out in two phases with the environmental and provisioning services commissioned in Phase 1 and cultural and regulatory services in a follow-on Phase 2.

Due to the urgency of the need for these evidence reviews, there was insufficient time for systematic reviews and therefore the reviews relied on the knowledge of the team of the peer reviewed and grey literature with some rapid additional checking of recent reports and papers. This limitation of the review process was clearly explained and understood by Defra. The review presented here is one of the ten evidence reviews which informed the IA.

Acknowledgments

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1 INTRODUCTION

The achievement of net-zero carbon emissions by 2050 is widely acknowledged to require an increase in the rate of carbon capture and sequestration at the national scale, in addition to significant reductions in emissions (Brown et al., 2021; Element Energy & UKCEH, 2021; Seddon et al., 2021). Commitments to reducing atmospheric CO_2 concentrations have been made by the UK government, under the Climate Change Act which commits the UK to achieving carbon neutrality by 2050 (Climate Change Act 2008) and an international pledge to the Paris Climate Agreement which commits the UK to limit global warming to well below 2°C (UNFCCC, 2015). A wide range of actions and initiatives have the potential to impact carbon sequestration, that could feasibly be implemented by land managers and farmers across the UK, including semi-natural habitat creation, changes to existing habitat management and changes to agricultural management practices. Specific land management objectives already introduced by the UK government include 12% woodland cover in England by 2050 (DEFRA, 2021c) including the planting of 30 000 ha of trees by 2024, and the banning of the sale of peat for use in amateur gardening by 2024 (DEFRA, 2021a). In this report, we reviewed the potential for a wide a range of land management options, provided by DEFRA, to result in carbon sequestration above or below ground. We define carbon sequestration as a net increase in long-term carbon stocks above and below ground, directly attributable to a mitigation activity. In this context, carbon sequestration may occur through increased rates of carbon removal from the atmosphere or through the protection of existing carbon stocks, over and above (i.e. *additional* to) expected rates in the absence of a given mitigation activity. As a result, land management practices may result in net carbon sequestration without producing a carbon sink in terms of absolute greenhouse gas (GHG) fluxes.

In this section of the report we

1) describe the scope of evidence and land management actions considered for review,
2) introduce the current status of carbon sequestration rates and stocks supported in the UK at a national scale,
3) highlight issues specific to below and above ground carbon stocks and sequestration, respectively and 4) discuss the variety of evidence that has contributed to the review and relationships between metrics.

<u>Scope of the report</u>

This review assesses the supporting evidence for all land management actions proposed by Defra that impact land use, land use change or forestry (LULUCF), which we judged could reasonably be expected to affect carbon sequestration. We report on evidence for carbon sequestration, detectible in either above or below ground carbon stocks, as supported by peer reviewed and grey literature evidence bases. Where present, evidence from England was given priority, however in many cases it was necessary to use evidence from the wider UK, Europe or beyond. Actions have been grouped into management bundles, in accordance with the type of management proposed. Areas of particular interest included (but were not limited to):

- Management of peatlands
- Increasing resilience to climate change
- Habitat creation
- Management of coastal habitats
- Management of woody vegetation
- Management of cropland
- Management of grassland
- Protection and maintenance of soils more generally

In most cases, evidence for carbon sequestration is reported on a per hectare basis. Estimates for potential carbon sequestration at a national scale may also be provided, typically for time horizon until 2050, where the area available for implementation has been estimated by primary sources. In addition to the assessing the evidence for carbon sequestration, we consider issues of spatial targeting, maintenance requirements and longevity, co-benefits and trade-offs for other ecosystems services, climate dependence, displacement and barriers to uptake.

Background

LULUCF is associated with significant fluxes of greenhouse gas emissions at a national scale, including both sinks and sources, and represents one of the most readily available options to enhance carbon sequestration (Element Energy & UKCEH, 2021; Gregg et al., 2021). A comprehensive assessment of current UK sequestration and emission rates from LULUCF (for managed land in the UK) is given by the UK greenhouse gas inventory report, which estimates total emission of 459.1 Mt CO_2-eq by the UK in 2019 (66% of emissions in 1990), including the net emission of 6.0 Mt CO_2-eq from LULUCF, where Mt CO_2-eq are megatonnes of CO_2 equivalents, including fluxes of CO_2, CH_4, N_2O, NO_x and CO (Brown et al., 2021). The overall flux estimated for LULUCF is the balance between net emissions from some land categories and net 'removals' (net carbon sequestration or carbon sink) in others, where a negative sign is indicative of carbon sequestration, and a positive sign represents net emissions to the atmosphere (Brown et al., 2021). In 2019, a net carbon sink of -16.3 MtCO_2-eq is estimated for Forest Land, with a further carbon sink associated with wood products harvested from UK Forest Land of -2.2 MtCO_2-eq. Grassland also contributes a net carbon sink, estimated at -1.1 MtCO_2-eq. The remaining GHG inventory land categories contribute net emissions of 15.9, 3.5 and 6.0 MtCO_2-eq., respectively for Cropland, Wetlands and Settlements (Brown et al., 2021). There is also a relatively small contribution from indirect N_2O emissions, related to atmospheric deposition of nitrogen, runoff of nitrogen originally applied as fertiliser etc. (0.2 MtCO_2-eq) (Brown et al., 2021). Within the UK greenhouse gas inventory, the emissions and removals contributing to different land categories include those arising from management is in those land categories, and also emissions and removals related to transitions between land use categories. An increase in national carbon sequestration through LULUCF can be achieved by a combination of:

- Protecting existing carbon stocks in soils and above-ground biomass.
- Enhancing the rate at which carbon is removed from the atmosphere by existing vegetation and stored semi-permanently in soils or biomass.
- Changing land cover through habitat creation to increase the rate at which carbon is removed from the atmosphere and stored semi permanently in soils or biomass.

The largest long-term carbon stocks in the UK from which losses could be prevented are found in soils, which have been estimated to contain 95% of UK carbon stocks, with the remaining 5% primarily stored in woodland biomass (Ostle et al., 2009). Carbon stocks in crop and grassland biomass are not considered to contribute to this total due to their lack of permanence. The greatest potential carbon sequestration rates (over decadal time scales) are observed in woodlands, followed by intertidal coastal systems (Gregg et al., 2021). Arguably, the greatest potential for increasing carbon sequestration is likely to lie in the creation of those habitats in areas currently associated with low carbon stocks and high emissions, such as arable land. However, land use change carries a significant risk of displacing prior land use activities (e.g. agriculture) either nationally or internationally. The emissions from displacement are typically not reported in accounts of habitat creation and associated carbon sequestration, and as such the risk of displacement does not directly contribute to the reported rates of carbon sequestration discussed below. However, displacement has the potential to significantly undermine local increases in carbon sequestration achieved through habitat creation (De Ruiter et al., 2016; Lambin & Meyfroidt, 2011; Meyfroidt et al., 2010; Roux et al., 2021). To ensure the role of displacement is captured, we instead addressed it under its own subheading and considered it as a separate indicator. Widespread changes in land management and land cover are also not without risks to carbon sequestration more locally and achieving the desired suite of ecosystem services is likely to require careful consideration of spatial targeting, maintenance requirements, and potentially complex co-benefits and trade-offs for other ecosystems services of value.

Below ground carbon stocks and sequestration

Carbon stocks in soils have the highest potential longevity of all carbon stocks in the UK but are susceptible to loss via erosion and elevated rates of decomposition following a change to environmental condition. Soil carbon stocks are the largest contributor to below ground carbon stocks at a national scale, which also includes carbon stores in root structures, litter and deadwood. Peatlands are particularly significant in terms of supported below ground carbon stocks but have become a source of significant GHG emissions nationally due to historic drainage and degradation or destruction of supporting habitats (Brown et al. 2021; Evans et al. 2017). Intertidal coastal sediments have also been identified as sites of significant below ground carbon stocks, as a result of deposition and in situ fixation (in the case of salt marsh) (Burden et al., 2019, 2020; Parker et al., 2021). The Countryside Survey is the most comprehensive assessment of topsoil carbon stocks in the UK with an average carbon concentration varying between 30.2 gC kg^{-1} in mineral soils and 467.9 gC kg^{-1} in organic soils in 2007, and reports a significant decrease in carbon concentrations and mean carbon density (tC ha^{-1}) in arable and horticultural topsoils from 1978 to 2007 (Emmett et al., 2010). The potential for carbon sequestration in arable soils, as a result of management, has been much debated in the UK. Whilst some research has suggested significant increases in soil carbon stocks could be achieved, more cautious assessments have highlighted issues with a lack of permanence in stocks, technical feasibility and soil carbon saturation (Chenu et al., 2019; Godde et al., 2020; Mattila et al., 2022; Poulton et al., 2018; Powlson et al., 2011, 2014; Smith et al., 2000). However, even low rates of net sequestration in arable soils have the potential to be nationally meaningful, due to the large areas available to implementation, the potential for added benefits to soil health and productivity, and for the sequestration of carbon without also displacing food and fibre production.

Significant knowledge gaps remain in our understanding of how land use and land cover change affect soil carbon at depth, and of how below ground carbon inputs from vegetation vary through space and over time. Analysis of the Countryside Survey data revealed that changes in in land use have a significant impact on topsoil carbon concentrations (Thomas et al., 2020). However, there is little evidence at the national scale for the response of total below ground carbon stocks to changes in land cover and land use. This is in part due to the technical challenge of surveying carbon stocks to depth at a national scale and the impact of short-term disturbance associated with the change in land cover itself (Luo et al., 2020). This knowledge gap is particularly relevant where land cover change is associated with a change in soil depth or a change in rooting depth (Jobbágy & Jackson, 2000; Stockmann et al., 2013). Any change in the distribution of litter inputs from above to below ground (or vice versa) adds to the difficulty of extrapolating from a change in topsoil carbon stock to full carbon stocks. In addition, the effect of soil carbon lost due to erosion on overall carbon cycling remains a source of significant uncertainty on the magnitude of emissions from UK agricultural soils (Lugato et al., 2018). Models of the relationships between above and below ground biomass production and carbon cycling are continuing to improve and provide insights into below ground vegetative carbon stocks and soil carbon stocks, but require accurate field measurements to constrain and validate predictions (de Gruijter et al., 2016; Luo et al., 2016; Paustian et al., 2016; Perugini et al., 2021).

Above ground carbon stocks and sequestration

Woodlands are the principal component of above ground carbon stocks and have been the subject of extensive study in the UK with respect to their carbon sequestration potential (Matthews et al. in prep.). It is well established that trees and woodland can support large carbon stocks above and below ground, along with high rates of carbon sequestration over policy relevant time scales (Matthews et al., in prep., Gregg et al. 2021; Poulton et al. 2003), however the number of studies reporting on vegetation carbon stocks and sequestration rates outside of woodlands is relatively limited in the UK (Gregg et al. 2021). Individual trees and hedgerows contribute significantly to national tree cover, but public data about the distribution of these features is limited (see Eddy, 2022; National Forest Inventory, 2017). Hedgerow extent is measured by the Countryside Survey, providing estimates of national coverage from a stratified random sample (Carey et al., 2008; Norton et al., 2012), but rates of sequestration are not reported. Research in woodlands has shown that potential rates of above ground carbon sequestration are highly dependent on vegetation management and species composition (Matthews et al. in prep.). More generally, woody vegetation has an important role in supporting a wide range of ecosystem services including biodiversity, flood protection and water quality, air quality, fibre production and cultural value (Brockerhoff et al., 2017; Chazdon, 2008; Ciccarese et al., 2012). In light on national commitments to tree planting (DEFRA, 2021c), uncertainties about the best types of trees to plant, where to plant them, methods of establishment and subsequent management objectives require resolution. Above ground carbon stocks also are vulnerable to a wide range of disturbances, including climate change, wildfire, and disease, in addition to competition with other land uses (IPCC, 2014). Many of these pressures are expected to intensify as a result of climate change, and questions have been raised about how to best safeguard above ground carbon stocks going forward (Allen et al., 2010; Hill et al., 2019; IPCC, 2014; Liu et al., 2010).

Interpreting metrics in the evidence

Across the evidence bases that contribute to this review, a variety of terms are used to report on rates of carbon sequestration. This is in part due to the lack of a consensus methodology and unequal research effort towards monitoring carbon fluxes across the wide range of habitats and management systems present in this review. Metrics quoted in this report include:

- rate of carbon sequestration (as the sum of multiple fluxes, see Equation 1)
- a specific flux into or out of carbon stocks (e.g. DOC loss, photosynthetic production)
- carbon stocks (repeat measurements of the carbon stock over time)
- a proxy for carbon stocks (vegetation area or height)

It is important to consider that changes in carbon stocks or sequestration rates are most often reported on a per area basis (standardised within the report to change per hectare (ha) where possible). However, the potential impact at the national scale is also dependent on the area available for implementation and feasibility. Although, we lack sufficient data to provide estimates of national sequestration in some cases (see Section 5: Evidence Gaps), we have included estimates of impact on carbon sequestration at a national scale where available, along with comments on potential barriers to uptake.

The overall impact of a change in land management on carbon sequestration is the sum of multiple fluxes occurring over different spatial scales and time frames, which are not consistently captured in the literature. As a result, reported magnitudes of carbon sequestration cannot be assumed to be directly comparable across the report. To demonstrate why this is the case, a summary of the fluxes that may or may not be included in assessments of net carbon sequestration are shown in equation 1. Assessments are typically limited to changes *in situ,* including changes in photosynthetic production, respiration, and decomposition. Deposition significantly contributes to local rates of carbon sequestration in riparian and coastal habitats but does not necessarily mean additional carbon has been sequestered at the national scale. Erosion is of particular importance in systems undergoing more intensive management but may go unreported in some assessments. Fewer assessments still account for the effects of management on:

- The fate of carbon manually removed from the site, including harvesting woody biomass
- Processing and management costs, including the burning of fossil fuels
- Displacement of land use activities

These fluxes can occur over different spatial scales and vary dramatically over different time frames, making complete reporting of all relevant fluxes challenging. Reported effect sizes are often highly sensitive to the duration and timing of a study, as in many habitats carbon stocks will respond rapidly to the initial change in management, before ultimately saturate at a new quasi-equilibrium value, potentially after many decades (Matthews, 2020; Poulton et al, 2003; Smith, 2014). Reported effect sizes can also vary from negative to positive over time, particularly where managements are associated with a high rate of initial disturbance, and the resulting carbon loss exceeds any increase sequestration in the short term (Matthews et al. in prep.).

As such, studies in the same system can appear to give markedly different estimate of sequestration potential, if carried out over different time periods or by accounting for different sets of process that affect carbon sequestration as a result of management change. Due to the potential importance of all these fluxes to the national potential for carbon sequestration, we have included accounts that incorporate all processes where possible. As stated previously, a negative sign is indicative of carbon sequestration, and a positive sign represents net emissions to the atmosphere (Brown et al., 2021). However, measurements of terrestrial carbon stocks are presented as positive values.

Equation 1 Fluxes that contribute to carbon sequestration. Fluxes that contribute gross sequestration (negative sign) are green, whereas those that contribute gross emission (positive sign) are purple. Note that a change in management could result in any individual flux increasing or decreasing in magnitude, relative to the existing baseline.

$$Carbon\ sequestration = -Production - Deposition + Respiration + Decomposition + Erosion + Removal + Processing\ Costs + Displacement$$

Lastly, when quantifying fluxes of greenhouse gases from terrestrial carbon stocks to the atmosphere, some studies account for only CO_2, whilst others also account for changes in the flux of methane (CH_4) or nitrous oxide (N_2O). Due to the wider objectives of carbon sequestration in the context of this report, (global, local, and regional climate regulation) we have incorporated discussions of all reported changes in greenhouse gas emissions in equation 2) where relevant, using units of carbon dioxide equivalence (CO_2-eq). To aid the interpretation of metrics reported below, we provide the conversion between units of tC and tCO_2eq in equation 2, both of which are used in this report.

Equation 2 Conversion from tC to tCO2eq

$$tCO_2eq = tC\,\frac{44}{12}$$

2 OUTCOMES

In this section we identify key findings from the list of actions reviewed, including which actions showed the most potential for carbon sequestration on a per hectare basis. We then highlight key, general issues arising from this review, which could hamper successful attempts to increase carbon sequestration capacity in England, including issues of displacement, the saturation of carbon stocks, carbon stock maintenance and longevity, data gaps and the feasibility of successful implementation. A further summary of key data gaps identified over the course of the report is provided in section 5.

Key findings from actions reviewed

This evidence review suggests that the largest potentials for carbon sequestration come from changes in land cover that involve taking land out of agricultural production, if managements are successful and follow the various caveats specified in the reviews below. In particular, the restoration of agricultural peatland and the establishment of woodland have a large potential for carbon sequestration in a policy relevant time frame. Changes in land management without a change in land use or land cover are unlikely to result in carbon sequestration to the same order of magnitude, on a per area basis. The potential exceptions to this rule are i) raising the average water table in agricultural land on peat, ii) changes in woodland management that promote an increase in average carbon stocks, and iii) managements that reduce the risk of carbon loss to pests, disease, erosion and wildfire. The likely impacts of pest, diseases and wildfire on carbon stocks over the next 30 years are difficult to estimate, but could significantly impact carbon sequestration long term if not accounted for in management schemes, and the risk of significant damage is also expected to increase as a result of climate change (IPCC, 2018). Also of note, is the potential to increase carbon stocks in arable soils through sustainable farm management. Managements in arable soils are associated with high uncertainty or smaller effects on carbon sequestration, on a per hectare basis, but the potential to impact carbon sequestration at a national scale is substantial, due to the large area available for implementation, assuming barriers to implementation can be overcome. Critically, increasing carbon stocks without taking land out of production will minimise displacement of emissions, which has the potential to undermine any carbon sequestration *in situ*. Key findings are summarised below, after which we discuss general issues which will affect the success of a wide range of actions.

- Agricultural displacement of food and/or fibre production (nationally or internationally) is a significant risk for a number of actions that have the potential to cause carbon sequestration. If displacement was to result in the creation of additional agricultural land in previously carbon rich habitats, then net carbon emission could occur as a result of actions that appear to have locally succeeded. Minimising the risk of harmful displacement is critical to carbon sequestration.
- Restoring degraded or wasted peatlands to peat-forming systems could result in the largest per hectare rate of carbon sequestration, but net impacts (sink plus avoided emissions) will be dependent on current rates of CO_2 emission, and risks of displacement.
- Where restoration is not possible on lowland peat, raising the water table and adapting farming systems accordingly is also likely to result in significant avoided carbon loss, although not a carbon sink.
- Planting trees and woodland can result in substantial carbon sequestration at national scales, but consideration of local context, any objectives in addition to carbon sequestration, future management and displacement risk are key.
- Preserving existing tree cover or increasing average tree cover by reducing management (harvesting) intensity can also increase carbon sequestration potential, but to a lesser extent. However, restricting harvesting can limit access to wood and biomass resources, which can impact GHG emissions in other sectors (such as energy and construction).
- Harvested wood can, under specific circumstances, contribute to long term carbon storage, and the avoidance of non-renewable materials and fuels, but intensifying forest management (harvesting)

through increased thinning or clearfelling innately involves carbon emissions and rapid loss of some carbon stocks.

- The restoration and creation of salt marsh in coastal areas with low above- and below-ground carbon stocks is likely to result in carbon sequestration. However, comparatively little is known about carbon dynamics in these systems, and the area of potential implementation is comparatively limited.
- Measures to increase below-ground carbon sequestration on agricultural land are typically associated with small increases in carbon sequestration on a per hectare basis, although the area available for implementation may make these actions nationally significant.
- Cover cropping, growing perennial crops, and incorporating ley rotations into arable systems have good potential for improving below ground carbon sequestration, mainly through increased inputs of organic carbon to soils. Increased SOC normally leads to improved soil physical structure, which is beneficial in itself, but may also lead to additional carbon sequestration through greater stabilisation of organic matter. Improved soil structure and greater vegetation cover also reduces soil erosion. There may be challenges incorporating these practices into arable rotations in England without displacing production.
- Both optimising soil pH (primarily through liming of acidic soils) and reducing fertiliser use can yield benefits to soil carbon sequestration, but both these interventions need to be properly targeted to avoid significant trade-offs such as reduced land productivity and substantial GHG emissions.
- Reducing tillage and no-tillage cultivation can help to sequester carbon in soils. However, effect sizes are associated with high uncertainty, with long-term effects argued to have been over-stated. The longevity of gains is also questionable, with carbon vulnerable to intermittent tillage that is often required by broader management objectives.
- The reduction of stocking rates for livestock is likely to benefit soil carbon stocks in managed grasslands, with evidence for small benefits on a per area basis. However, information specific to the UK is somewhat lacking and management outcome highly sensitive to effective targeting and the avoidance of displacement.
- Agroforestry has the potential to significantly increase carbon stocks on agricultural land, however there is a need for careful targeting to minimise initial below ground carbon loss and risks of displacement.
- For all actions, monitoring changes in carbon stocks long-term to ensure service delivery and inform management refinements is highly advisable, particularly under changing climatic conditions and with the adoption of new technologies and crops.
- Some interventions are likely to result in a short-term loss of below ground carbon stocks that will ultimately be offset by substantial long-term gains. This should be taken into account when projecting the impacts of managements on carbon stocks for policy relevant time frames, and initial carbon losses minimised where possible.

Displacement

Despite having the largest potential for increasing carbon sequestration, actions which remove land from food or fibre production are likely to cause displacement, without an associated reduction in demand. Displacement can occur as an increase in management intensity or an increase in the area under production outside of land management schemes, to compensate for losses as a result of one of the actions reviewed below. Both processes can cause the emission of GHGs, to the extent that local carbon sequestration measures ultimately result in net emissions at a national or global scale (De Ruiter et al., 2016; Lambin & Meyfroidt, 2011; Meyfroidt et al., 2010; Roux et al., 2021). The international displacement of agricultural production carries particular risks for offsetting carbon sequestration, where carbon dense habitats may be converted to agricultural production as a result (Gibbs et al., 2010; Pan et al., 2011; Pendrill et al., 2019). In the case of tropical deforestation for agricultural production, the loss of carbon stocks, sequestration potential and loss of biodiversity greatly outweigh those that that could potentially be generated in the UK.

Beyond agricultural displacement, land is also required to support other ecosystem services, beyond carbon sequestration and food and fibre production. Managements that promote carbon sequestration will variably align with or oppose the generation of other desirable ecosystem services, such as flood protection, air quality, biodiversity and cultural services. Not only is there limited space available for all of these services to be generated, but the extent to which they may be generated in concert will be highly context dependent. Appropriate spatial targeting of managements, clear local management objectives, and access to suitable specialist support for land managers will be important to ensure successful outcomes for ecosystems service provision at local and national scales.

<u>Saturation of carbon stocks</u>

Rates of carbon sequestration that can be achieved over the next three decades will be critically important for efforts to reach net zero by 2050, but sustained rates of sequestration will also be necessary to continue to offset emissions beyond that point. Across many habitats, there is evidence for initially rapid rates of sequestration following a change in management, particularly where carbon stocks have been historically degraded (Burden et al., 2019; Element Energy & UKCEH, 2021; Smith, 2014). However, rates of sequestration will gradually slow, as rates of respiration and decomposition increase (Poulton et al., 2018). Ultimately, this can lead to the saturation of carbon stocks, and the need for additional methods of carbon sequestration to offset any further anthropogenic emissions, without jeopardising existing stocks (Element Energy & UKCEH, 2021). In general, we have a good understanding of the long-term trajectory for sequestration in woody carbon stocks, including the point at which above ground carbon stocks will saturate, and the dependence of sequestration rates on management and species productivity (Matthews 2020; Matthews et al, in prep.). The dynamics of soil carbon saturation are more challenging to predict, due to the innate variability of soils and complex relationships between carbon pools. The saturation of carbon stocks in mineral associated organic matter, which is more stable than other soil carbon fractions and considered key to long-term carbon sequestration below ground, is a well-established concept (Chen et al., 2019; Nicoloso et al., 2018; Paterson et al., 2021). Beyond this, the saturation point of total soil carbon stocks (including particulate and free organic carbon) is a product of soil properties, rates of organic matter inputs and environmental conditions, including temperature, hydrology, nutrient availability, pH and disturbance (Cotrufo et al., 2015; Paterson et al., 2021; Possinger et al., 2020). The potential for carbon sequestration in mineral agricultural soils will depend on the magnitude of historic degradation and current management practices (Georgio et al., 2022), but stocks are ultimately expected to saturate (Poulton et al., 2018; Smith, 2014). Where high concentrations of organic matter can be supported below ground with comparatively low rates of decomposition (e.g. peatlands), carbon may continue to accumulate long term, although rates of accumulation are thought to be low and vulnerable to disturbance events (Young et al., 2021). Although managements to increase semi-natural carbon stocks could play an important role in limiting climate change over the next century, we emphasise the eventual saturation of these carbon stocks to showcase the importance of reducing absolute rates of greenhouse gas emissions, from LULUCF and other sectors, to mitigate climate change long term.

<u>Data gaps</u>

Over the course of this review, we have reported on several interventions with the potential to result in significant carbon sequestration, but where evidence is either spatially restricted, high levels of variability exist between studies, or where sequestration is dependent on the assumption of successful restoration, the determinants of which as less clear. These evidence gaps provide a strong case for the active monitoring of carbon sequestration rates in concert with the implementation of actions considered below. In addition to providing a valuable evidence base for land managers, scientists and policy makers to refine managements and future management recommendations, it would provide an opportunity for validation and the implementation of outcome-based payments, which may help prevent perverse outcomes (Hejnowicz et al., 2014; Reed et al., 2014).

An example of a promising intervention associated with notable data gaps is the natural regeneration of woody vegetation (Broughton et al., 2021; Lewis et al., 2019; Poulton et al., 2003, see review below). In the

UK, natural regeneration has been shown to result in significant carbon sequestration in addition to supplying other ecosystem services for low initial and maintenance costs (Poulton et al., 2003; Matthews et al., in prep.). However, our ability to predict the success of natural regeneration in the UK as a whole is limited, due to a low number of study sites in the UK and a strong dependence of outcomes on local conditions. As such, monitoring of natural regeneration, where implemented, to inform appropriate management is advised. Additionally, public information about the existing distribution of woody vegetation below a minimum parcel size of 0.5 ha in rural environments (e.g. hedgerows) is lacking, it is not possible to anticipate the full capacity for natural regeneration at a national level. The advancement of remote sensing capabilities towards high resolution habitat mapping and biomass estimations has the potential to significantly shrink these knowledge gaps for above ground biomass and sequestration potential if deployed at a national scale if supported by adequate ground truthing (Gray et al., 2018; Jeronimo et al., 2018; Räsänen et al., 2019; Wilkes et al., 2018). We discuss some of these issues further in section 5.

Feasibility

Over the course of this review, multiple actions were identified where technological, logistic, economic or cultural barriers could hinder uptake or successful implementation. The actions reviewed vary from expanding tried and tested managements to implementing novel practices in a UK context. The likelihood of success when implementing more novel practices at scale, such as paludiculture (wet agriculture and/or forestry on peatlands), is largely unknown. Such actions would benefit from further study and trials in a commercial context (where relevant) before being deployed at a national scale. Some interventions that are likely to be most effective for carbon sequestration, including the rewetting of both upland and lowland peat and the establishment of large woodland (in the absence of displacement), also face significant logistical issues, as actions may require coordinated action across multiple land owners and water boards. Additional support structures may be required to realise the full potential of actions operating at larger spatial scales, and to facilitate collaboration across landowners. Furthermore, access to specialist advice and the ability to monitor outcomes, will be particularly important where barriers to success are more likely. Finally, in addition to likely impacts on hydrology and biodiversity considered in other sections of this report, widespread changes in land cover and habitat creation will impact cultural services and the visual quality of landscapes in diverse ways, generating a range of responses in the wider public (Nijnik et al., 2009). This is in addition to challenges to uptake that arise from existing reluctance amongst landowners.

3 MANAGEMENT BUNDLES

The actions fully assessed in this report for their influence on carbon sequestration are presented grouped in the following management bundles and sub-bundles:

1. **Actions for habitats with specific hydrological features**
   i. Floodplains
      a. **ETPW-016C**: Create water meadows
      b. **ETPW-036EM**: Enhance/ manage floodplain meadows
   ii. Peatlands and Wetlands
      c. **EBHE-164C**: Create wetland habitat
      d. **ECAR-041**: Reduce managed burning on non-SAC/SPA designated sites and on shallow peat
      e. **ECCM-030**: Restore/manage upland and lowland peatlands including blanket bog and raised bog
      f. **ECCM-034**: Remove non-peat habitat vegetation
      g. **EHAZ-137**: Use paludiculture
      h. **ECCM-038**: Raise water levels in areas of farmed peatland and adapt farming systems accordingly
      i. **ETPW-153**: Stabilise eroding peat through targeted restoration work

2. **Climate measures**
   i. Climate change mitigation and adaptation
      a. **ECCA-005**: Undertake a climate change vulnerability assessment
      b. **ECCA-026**: Plant a range of native species, including trees grown from locally adapted and genetically diverse seed sources, and from more southerly provenances
      c. **ECCM-074**: Plant, enhance or manage bioenergy crops (e.g. short rotation coppice)
      d. **ECCM-065**: Switch to using peat alternatives in horticultural growing media

3. **Habitat creation**
   i. Agroforestry
      a. Create agroforestry systems [**ECAR-032**]
   ii. Coastal
      a. Create sand dunes [**EHAZ-070C**]
      b. Create shingle features [**ETPW-179C**]
      c. Create inter-tidal and saline habitats [**ETPW-180C**]
      d. Create saltmarsh [**ETPW-180C**]
   iii. Grassland
      a. Create species rich grassland habitats [**ECPW-022C**]
   iv. Hedgerow
      a. Plant hedgerows [**ECCM-025C**]
   v. Mountain, moor and heathland
      a. Create heathland (including heathland mosaics) [**ECPW-176C**]
   vi. Scrub
      a. Create targeted scrub [**EBHE-203C**]
   vii. Woodland
      a. Create a woodland creation plan [**EBHE-104**]
      b. Set up or engage with community tree planting projects [**EBHE-281**]
      c. Create woodland on a large scale [**ECCM-048**]
      d. Create floodplain woodland [**ECPW-071C**];
      e. Create traditional orchards with local varieties of fruit tree [**EBHE-209C**]
   viii. Woody features
      a. Plant or manage trees outside of woodlands, including shelterbelts [**ECCM-024**]

4. **Litter and waste**
      a. Use of biodegradable silage, crop cover mulches and planting trays to meet recognised compostable standard EN17033 [**ECPW-281**] & Use alternatives to fossil-based plastic mulches, such as green mulches or other

biodegradable materials. Straw, shredded wood and other natural products can also be used as mulch [**ECPW-280**]

5. **Monitor, plans, databases, consultation and resulting action**
   a. Monitor health of trees [**ECCM-058**]

6. **Natural regeneration**
   a. Create woodland through natural regeneration [**ECCM-049**]
   b. Allow natural regeneration and extension of existing habitat (e.g. hedgerows, scrub, rough grassland)[**ETPW-171**]

7. **Restoration, management and enhancement**
   i. Coastal
      a. Active management of coastal realignment [**EHAZ-074**]
      b. Enhance / maintain sand dunes [**EHAZ-070EM**]
      c. Enhance/ manage inter-tidal and saline habitats [**ETPW-180EM**]
      d. Enhance manage salt marsh [**ETPW-081EMX**]
      e. Enhance/ manage coastal habitats [**ETPW-081EM**] is reviewed across the section.
   ii. Cropland
      a. **ECCM-021**: Farm perennial crops
   iii. Grassland
      a. Reduce stocking rate (grazing) to restore structure and flowering, maintain ground cover, and reduce poaching [**ETPW-104**]
      b. Use herbal and grass leys [**ECPW-032**]
      c. Enhance and manage locally distinctive flower rich/hay meadows using traditional techniques [**EBHE-214EM**]
   iv. Hedgerows
      a. Enhance/ manage hedgerows [**ECCM-025EM**]
   v. Mountain moor and heath
      a. Enhance or manage moorland (including common land), e.g. through appropriate traditional grazing techniques
   vi. Riparian
      a. **ECPW-291C** Create riparian habitats
   vii. Woody features
      a. Enhance/ manage wood pasture (e.g. through appropriate grazing) [**EBHE-205EM**]
      b. Manage veteran and ancient trees [**ECCM-056**]

8. **Soil management and protection**
   i. Compaction management
      a. **ETPW-223**: Assess soil structure and plan how to avoid and alleviate soil damage and compaction (soil management plan)

9. **System actions**
   i. Systems action – Mixed systems and cross habitat action
      a. **ECCA-035** Prepare and implement wildfire management plans
   ii. Systems action – Pests and disease management
      b. **ETPW-265** Fell diseased trees where the action is uneconomic

3.1 ACTIONS FOR HABITATS WITH SPECIFIC HYDROLOGICAL FEATURES

This bundle concerns actions that create or manage wetland habitats, distinguishing between floodplains, most of which have mineral soils, and other wetland habitats, the majority of which are peatlands (Butcher et al., 2020). Peatlands cover an approximate 3 million hectares in the UK (Evans et al., 2017) and store over

3000Mt of sequestered carbon as organic carbon within sediments (Lindsay, 2010). The preservation and restoration of these sediments has become a priority for research and UK policy as a result of observed rates of GHG emissions from degraded peatland (Evans et al., 2017). In the context of carbon sequestration, we report on actions associated with the creation and management of floodplains, the restoration of peatlands and management of agricultural peatlands.

The main outcomes of this review were that:

- Restoration of degraded peatland has a strong supporting evidence base and is actively being pursued in the UK.
- Lowland agricultural peats have the highest emissions abatement potential from restoration, but have competing demands for food production.
- The adaptation of farming systems has the potential to resolve this conflict, with some promising evidence emerging, but the development these methods in the UK is not sufficiently advanced sufficiently for a full assessment.
- There is a risk of elevated methane emissions following restoration of peatlands, but even accounting for these potential changes, high rates of net emissions abatement are expected from peatland restoration in the majority of cases.

3.2 Actions for habitats with specific hydrological FEATURES - Floodplains

3.2.1 ETPW-016C: Create water meadows

3.2.1.1 Causality

Food and fibre production	Area under production or management intensity outside of ELM	T*
Global, regional & local climate regulation	Above ground carbon sequestration	LT*
	Below ground carbon sequestration	LT**

The benefits of creating water meadows are likely to be significant if creation involves restoring floodplain from more intensive agriculture, but information about rates of sequestration in floodplain meadow are lacking.

Studies of floodplain meadows in England and Wales show inconclusive rates of in situ carbon fixation and sequestration, and further research is necessary to quantify the relevant fluxes robustly (Rothero et al., 2016). As such, creating floodplain meadows specifically cannot be said to offer significant carbon sequestration benefits (Lawson et al., 2018), although when replacing arable land an increase in sequestration rate is highly likely (Gregg et al., 2021).
Where the creation of floodplain meadows involves the hydrological restoration of the floodplain, there is good evidence from sites across Europe that significant local carbon sequestration will occur in sediments (Gregg et al., 2021). Approximately 42% of floodplains in England and Wales are no longer connected to river systems (Gregg et al., 2021), and high rates of intensive agricultural land use occur on floodplains in the UK which has a strong negative impact on carbon storage in floodplain soils (Gregg et al., 2021).
Following hydrological restoration, initial rates or carbon sequestration can be particularly high, with sequestration rates of -1 tC ha^{-1} yr^{-1} estimated for the first century following restoration along parts of the Danube in Austria, dropping to 0.08-0.18 tC $ha^{-}1yr^{-1}$ at time scales of 300 to 600 years (Zehetner et al. 2009). Although rates sequestration decrease exponentially, carbon stocks in floodplains can be relatively high, compared to other terrestrial habitats, with 109.4 t C ha^{-1} found in the top 10 cm of soil in Cricklade National Nature Reserve (Gregg et al., 2021). More widely, estimated values for temperate floodplain carbon storage are on the orders of magnitude of 100-1000 tC ha^{-1} with reported accumulation rates of -0.1 to -3 tC ha^{-1} yr^{-}

^{1} (although the majority of accumulation rates are less than -1 tC ha^{-1} yr^{-1}), which are affected by climate, geology, land cover, and fluvial properties (Sutfin et al., 2016).

However, it is important to consider that a significant proportion of carbon sequestration in floodplains is allocthanous particulate organic carbon (carbon that was not sequestered from the atmosphere at its current location and has been transported) that is deposited when floodplains are inundated (Cook, 2007). This means that the increase or decrease of below ground carbon in floodplains is not a reliable indicator of carbon gain or loss at the landscape scale, as reduced deposition may be due to improved erosion protection upstream, and increased erosion does not necessarily mean carbon has not been sequestered downstream, for example in saltmarsh. As such, the additionally of floodplain carbon stores is associated with a large amount of uncertainty (Quinton et al., 2010).

Lastly, significant methane emissions have be documented in wetlands due to microbial respiration in anoxic conditions, which may offset some of the benefits form wetland restoration and reduced agricultural emissions (IPCC Task Force on National Greenhouse Gas Inventories, 2014).

3.2.1.2 Co-Benefits and Trade-offs

No assessment.

3.2.1.3 Magnitude

Long-term, storage of carbon in floodplains can vary on the order of magnitude between 100-1000 t C ha^{-1} (sample depths are unknown) with an accumulation rate of <-0.1 – -3 tC ha^{-1} yr^{-1} (Sutfin et al., 2016), although higher rates of sequestration may be achieved following hydrological restoration of floodplains. Floodplains cover an estimate 1.6 million ha across England and Wales, with 42% no longer connected to river systems (Gregg et al., 2021).
More specific sequestration rates on floodplain meadow, relative to other semi-natural or managed floodplain systems are unknown. Any emissions resulting from or prevented by the conversion of land cover should also be considered.

3.2.1.4 Timescale

>10 years – to accumulate carbon stores through sequestration and deposition.
Establishment of floodplain meadow dynamics and representative components of the floristic community will be faster (The River Restoration Centre, 2020).

3.2.1.5 Spatial Issues

Collaborative work required to be effective (this may be combined with targeting)
Downstream issues are critical e.g. impacts on coastal systems.
The impacts of creating a floodplain on fluvial dynamics and regional flood risk should be considered.
If creating a floodplain meadow on an existing floodplain, concerns would be less significant.

3.2.1.6 Displacement

Arable activities and some grazing activities may be displaced if floodplain meadow is replacing intensely grazed or cultivated floodplain.

3.2.1.7 Maintenance and Longevity

Active management of the water table, grazing pressure and cutting are typical managements of floodplain meadow that would be required in perpetuity, to preserve the system (Rothero et al., 2016).

3.2.1.8 Climate Adaptation or Mitigation

Additional sequestered carbon would constitute climate change mitigation.

3.2.1.9 Climate Factors / Constraints

Given the specific hydrological regimes required for the maintenance of floodplain meadows, increasing climatic variability, specifically flood and drought risk may necessitate more intensive hydrological regulation.

3.2.1.10 Benefits and Trade-offs to Farmer/Land manager

There may be a loss of land for agricultural activities if arable land is lost or grazing intensity reduced to create the floodplain meadow.
There may be a tourism or amenity benefit, as floodplain meadows receive high numbers of visitors on an annual basis. Landowners may also benefit from reduced flood risk on neighbouring assets.
Continued management of the meadow will be an ongoing expense, in addition to conversion costs. Floodplain meadows often receive designated status and constraints upon future activities (Rothero et al., 2016).

3.2.1.11 Uptake

Flood plain restoration and creation is occurring in the UK on a small scale. Identifying appropriate sites with incentives to reduce agricultural pressures may be the main challenge. Improving the evidence base for our understanding of carbon dynamics in floodplain meadows is also highly desirable.

3.2.1.12 Other Notes

None

3.2.2 ETPW-036EM: Enhance/ manage floodplain meadows

Duplicate evidence:

3.2.2.1 Causality

Food and fibre production	Area under production or management intensity outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**L***
	Below ground carbon sequestration	**L***

Studies of floodplain meadows in England and Wales show inconclusive rates of carbon sequestration, and suggest that further research is necessary to quantify biogeochemical fluxes robustly (Rothero et al., 2016). As such, floodplain meadows cannot be said to offer significant carbon sequestration benefits (Lawson et al., 2018), and the magnitude of any effects from management or enhancement for carbon are unknown. For comments on the carbon sequestration potential of floodplains in general, see " ETPW-016C: Create water meadows".

Managements carried out on floodplain meadows are generally associated with greater rates of carbon sequestration, using evidence bases from other habitats, but quantitative evidence from floodplain meadows specifically is lacking. These managements include maintaining water levels and managing grazing intensity or cutting frequency (Lawson et al., 2018; Rothero et al., 2016). Evidence from other habitats (Abdalla et al., 2018) and logic chains suggest that low grazing intensity is likely to confer a positive impact on the preservation of carbon stores, by minimising compaction and erosion, particularly where there is evidence of poaching. Floodplain meadows are regularly drained to ensure that the habitat is not inundated for too long, resulting in changes to the botanical community (Rothero et al., 2016). The effects of intermittent drying and drainage on floodplain carbon fluxes is unknown, but studies of drained peatland and mineral soils suggest it may increase rates of carbon loss on soils with high organic matter content (Evans et al., 2017). As discussed in "ETPW-016C: Create water meadows", the additionally of floodplain carbon stores is associated with a large amount of uncertainty, as carbon accumulation in floodplains is variably due to the storage of carbon relocated from upstream, and not *in situ* carbon sequestration (Quinton et al., 2010).

3.2.2.2 Co-Benefits and Trade-offs

[TOCB Report-3-3 Soil] Actions that create wetter lowland environments in which livestock graze can have potential limited disbenefits for the extent of erosion and soil structure due to exposure of bare soil and poaching. However, the level of risk will be highly context dependent (e.g. if a created wetland is fenced off to restrict livestock access the risk to soil structure and erosion will be eliminated). In addition, the wetter environments are also likely to increase soil organic matter contents thereby benefitting below ground C sequestration and some aspects of soil health.

[TOCB Report-5B Grassland]: Maintaining high water levels will benefit existing wetland habitats and species. The effect of restoring high water levels could provide new wetland habitats but where these replace well-established semi-natural vegetation the effect could be detrimental to some species.

3.2.2.3 Magnitude

Unknown.

3.2.2.4 Timescale

Unknown.

3.2.2.5 Spatial Issues

Limited to current locations and constrained by surrounding land. Implementation might be restricted as a result of low water availability, or if management is associated with high levels of nutrient release.

3.2.2.6 Displacement

Reducing grazing pressure may displace grazing activities to other habitats.
Drainage to maintain desired hydrological regimes may displace water downstream from a potential floodwater storage site.

3.2.2.7 Maintenance and Longevity

Active management of the water table, grazing pressure and cutting are typical managements of floodplain meadow that would be required in perpetuity, to preserve the system (Rothero et al., 2016)

3.2.2.8 Climate Adaptation or Mitigation

Additional carbon sequestered *in situ* would constitute climate change mitigation. Any long-term storage of carbon transported from elsewhere might also contribute to a reduction in emissions, but this is highly uncertain and not easily quantifiable.

3.2.2.9 Climate Factors / Constraints

Given the specific hydrological regimes required for the maintenance of floodplain meadows, increasing climatic variability, specifically flood and drought risk may necessitate more intensive hydrological regulation.

3.2.2.10 Benefits and Trade-offs to Farmer/Land manager

There may be a tourism or amenity benefit, as floodplain meadows receive high numbers of visitors on an annual basis. Landowners may also benefit from reduced flood risk on neighbouring assets. However, this is likely to result from general maintenance of floodplain meadows, rather than enhancement for carbon storage.

Continued management of the meadow will be an ongoing expense, in addition to conversion costs. Floodplain meadows often receive designated status and constraints upon future activities (Rothero et al., 2016).

3.2.2.11 Uptake

Flood plain restoration and creation is occurring in the UK on a small scale. Identifying appropriate sites with incentives to reduce agricultural pressures may be the main challenge. Improving the evidence base for our understanding of carbon dynamics in floodplain meadows is also highly desirable.

3.2.2.12 Other Notes

The observed increase or decrease of carbon in floodplain sediments is not a reliable indicator of carbon sequestration or emission at the landscape scale, as reduced local deposition may be due to improved erosion protections upstream, and increased local erosion does not necessarily mean carbon has not been sequestered downstream. As such, the additionally of floodplain carbon stores has a large uncertainty associated with it, beyond any uncertainty in the expected magnitude of local carbon accumulation.

3.3 Actions for habitats with specific hydrological features – Peatlands and wetlands

3.3.1 EBHE-164C: Create wetland habitat

Duplicated evidence base for: Create wetland habitats and mosaics, including creating the appropriate hydrological conditions [**ECCA-007C**]; Create artificial wetlands [**ECCA-013C**]; Create fen [**EHAZ-129C**].

3.3.1.1 Causality

Food and fibre production	Area under production or management intensity outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**LT***
	Below ground carbon sequestration	**LT***

There has been little research into the possibility of establishing novel inland wetlands with the intention of sequestering significant volumes of carbon (as opposed to restoring historic wetland in various states of degradation – see section 3.3.3). Creating new wetland may have additional benefits due to the abatement of emissions associated with prior land use, but would not confer the significant rates of emissions abatement that can be achieved by restoring existing, degraded peat, which emits an estimated 18.8 MtCO_2-eq yr^{-1} (Brown et al., 2021). Long term average rates of carbon sequestration in peat are low, but relatively little is known about rates of sequestration that could be achieved in the shorts term, on policy relevant time scales, although modelling of carbon sequestration rates following peatland restoration suggests that more rapid rates of sequestration could occur early on (Element Energy & UKCEH, 2021).

The majority of wetland habitats in the UK are peatland, which in a near-natural state significant amounts carbon (gross flux) on an annual basis (-3.5 to -5.4 t CO_2 ha^{-1} yr^{-1}), but are approximately net neutral with respect to GHG emissions on an annual basis (based on 100 year Global Warming Potentials, GWP_{100}) as a result of emissions of CH_4, and to a lesser extent CO_2 emitted from the decomposition of exported dissolved organic matter, although exact rates and uncertainties vary between bog and fen (Evans et al., 2017). Over longer time horizons peatlands are strongly net cooling, because the emitted CH_4 has a short (decadal) atmospheric lifetime, whereas the sequestered CO_2 remains securely stored in the peat over millennia. Inland, freshwater wetlands on mineral soils are thought to be less important for carbon sequestration, although past conflations of the evidence base from peatlands with that of all wetlands in some sources appears to confuse this issue (Element Energy & UKCEH, 2021). The development of peat deposits in natural wetland occurs over millennia following the establishment of a high water table. Establishing new wetlands would require the artificial raising of the water table, which is a method being employed in current wetland restoration projects (The Great Fen Project, 2010), but which could provide significant challenges at scale. Long term rates of peat formation are generally thought to be too slow to be of significance to climate change mitigation on decadal time scales, but recent work suggests that rapid rates of peat formation can occur during the early stages of creation due to lower rates of decomposition than in older peats (Element Energy & UKCEH, 2021; Young et al., 2021). Although managements for accelerated peat formation, by enhancing net primary productivity (NPP) have been demonstrated experimentally, it has not been tested at scale or outside of the context of wetland restoration (Element Energy & UKCEH, 2021).

3.3.1.2 Co-Benefits and Trade-offs

No assessment.

3.3.1.3 Magnitude

Brown et al. (2021) report that near natural bogs sequester -3.54 $tCO_2\ ha^{-1}yr^{-1}$, whilst near natural fens sequester -5.41 $tCO_2\ ha^{-1}yr^{-1}$ (both values excluding estimated loss of 0.69 $tCO_2\ ha^{-1}yr^{-1}$ as DOC). Including all reported GHG fluxes, near natural bogs have an emissions factor of -0.02 tCO_2-eq $ha^{-1}yr^{-1}$ and near natural fens have and emissions factor of -0.93 tCO_2-eq $ha^{-1}yr^{-1}$.

However, recent research has suggested that higher rates of sequestration could be achieved early in the establishment phase (Young et al., 2021). Based on this research, Element Energy & UKCEH (2021) estimate that an average sequestration rate of -8.6 $tCO_2\ ha^{-1}\ yr^{-1}$ could be achieved over a 50 year period from effective peat restoration, excluding net sequestration from avoided emissions (an additional -10 to -15 $MtCO_2$-eq yr^{-1}) that can only be achieved through restoration rather than creation of new wetland. Methane emissions of approximately 3 tCO_2-eq $ha^{-1}yr^{-1}$ are also predicted. However, evidence for this rapid early accumulation of carbon stocks in peat is very limited, and additional empirical support is needed.

3.3.1.4 Timescale

> 10 years
Establishment of hydrological function can take decades, although appropriate vegetation can establish between 5-10 years, using peatland restoration as a guide. Accrual of significant carbon stocks in peatland occurs over millennia.

3.3.1.5 Spatial Issues

The creation of wetland habitat is dependent on the provision of a sufficiently high water table all year round.

3.3.1.6 Displacement

This action would likely displace agricultural activity.

3.3.1.7 Maintenance and Longevity

Long term maintenance would be required to prevent drying and emissions of accumulating carbon stocks.

3.3.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.3.1.9 Climate Factors / Constraints

All wetlands are expected to be vulnerable to increasing drought risk and competition for water resources due to climate change.

3.3.1.10 Benefits and Trade-offs to Farmer/Land-manager

May benefit from carbon credits and could help moderate hydrological extremes in surface waters through water storage capacity to a small degree, hence alleviating flood risks.

3.3.1.11 Uptake

Potentially reduces area of productive land.

3.3.1.12 Other Notes

Although creating wetland may have some carbon benefit, restoration of degraded peat has a high likelihood of providing much greater return in emissions abatement for largely the same management investment.

3.3.2 ECAR-041: Reduce managed burning on non-SAC/SPA designated sites and on shallow peat

3.3.2.1 Causality

Global, regional & local climate regulation	Above ground carbon sequestration	LD*
	Below ground carbon sequestration	**LD***

Evidence of the impact of controlled burning on SOC stored and net sequestration is unclear on all peat depths, including 'shallow' peats, defined in England as where the thickness of the surface horizon is between 10 and 40 cm (Joint Nature Conservation Committee, 2011). A previous report on the impacts of controlled burning in the Welsh uplands reported that effects in the literature were unclear, with studies disagreeing in both the magnitude and direction of effect (Alison et al., 2019). The majority of studies compare the effects of burning with no burning on peatland, with less evidence for the effects of reduced frequency burning, and variation in the response metrics used across studies.

The exact area affected by burning on shallow peats is not well known. The area of shallow peats in the UK is approximately 4.7 million ha, 74% of which occurs is Scotland, acknowledging that the definition of shallow peats varies across nations (Joint Nature Conservation Committee, 2011). In England, there are an estimated 527,193 ha of shallow peat (4%). The purpose behind managed burning in upland moors is variable, and reasons include the managed reduction of the severity and frequency of wildfire, to preserve or manage a specific vegetation community, and for game bird management (Alison et al., 2019; Davies et al., 2016)

Evidence for the impact of managed burning on organic carbon rich soils is limited, with no evidence found for shallow peats, and no evidence explicitly for SACs or SPAs. As a result, the evidence presented below is associated with a surface horizon >40cm deep, and the degree to which these results will translate to shallow peats is not known. Burning has been reported to predominantly favour heather (*Calluna*) over blanket bog species (*Sphagnum*) in muir burning, depending on burn cycle length (Alison et al., 2019; Worrall et al., 2011), which suggests a detrimental effect on peat forming vegetation that may negatively affect SOC formation. Negative effects of burning on carbon accumulation were found in the only long term manipulation experiment in the UK (Garnett et al., 2000; Marrs et al., 2018) as reported by (Alison et al., 2019). Harper et al. (2018) report that the effect of burning on soil carbon is typically negative, although highlight a general lack of evidence. Studies of regular burning on peat have been associated with *Sphagnum* and cotton grass cover by up to 5 times that of unburnt plots at multiple sites within the UK (Whitehead et al., 2021), however unburned sites were in heather dominated moorland with a history of burning rather than near-natural bog, where burning will not support sphagnum cover (Baird et al., 2019). The effects of burning on carbon losses via DOC are also not well established in the literature, due to a predominance of observational studies over experimental ones (Evans et al., 2017).

3.3.2.2 Co-benefits and Trade- offs

Reducing the frequency or intensity of managed burns is suggested to increase the risk of wildfire in the absence of peatland restoration. Interactions between managed burns and wildfires in peatlands is complex (Davies et al., 2016) and burning for wildfire risk management remains contentious (Gregg et al., 2021). Under drought conditions, or where peats are subject to drainage wildfires risk igniting the peat itself, in addition to damaging peat forming vegetation, with potential for large carbon loss and pollution generation (Davies et al., 2013; Turetsky et al., 2015). The risks for large carbon loss and pollution are likely to be smaller on shallow peats that those typically reported for peat, but potentially still significant. Emissions from historic peatland wildfires in the UK have been highly variable (Gray et al., 2021). However, hydrological restoration of peatland in addition to cessation of burning would likely be a more effective method of reducing fire risk whilst delivering additional emissions abatement (Gregg et al., 2021), and there is no evidence that peatlands in a near natural state are a wildfire risk (Baird et al., 2019). Once again, most of this discussion has developed concerning peats where the surface horizon is > 40cm deep.

3.3.2.3 Magnitude

Marrs et al. (2018) found that prescribed burning on peatland did not prevent net carbon accumulation, but that increasing burn frequency reduced the rate of carbon sequestration by 1.9 gC $cm^{-2}yr^{-1}$ (0.019 tC ha^{-1} yr^{-1}).

3.3.2.4 Timescale

0-5 years
Burning has an immediate effect on carbon emissions and NPP. However, longer term consequences of burning are harder to establish, with some effects (such as increased dominance of fire-prone *Calluna*) developing over many years.

3.3.2.5 Spatial Issues

The vulnerability of peatlands to burning is likely to vary across the UK and with the level of degradation. Peatlands that have been drained or degraded are likely to have a greater vulnerability to wildfire. When peatlands are in a near-natural state, the risk of wildfire is significantly reduced and restoring peatland by raising the water table and blocking drainage may confer similar protections. The evidence base for the impacts of burning on peatland is also biased towards England, specifically the Pennines.

3.3.2.6 Displacement

If reduced burning prevents activities such as grouse rearing at the site of management these activities and associated burning may be displaced elsewhere, although this activity is highly unlikely to move from the uplands into productive agricultural land. Furthermore, the evidence that grouse numbers would be lower under a no-burn regime (e.g. mowing) is not clear, so displacement may not be inevitable. The consequences of any displacement would depend on the relative vulnerability of the sites to burning.

3.3.2.7 Maintenance and Longevity

Any re-establishment of a burning regime or increase in frequency in the future would be assumed to reverse any changes due to reduced burning, with potentially more intense fires at the reintroduction of burning practices. On the other hand, successful transition of a dry, *Calluna*-dominated blanket bog to a wet, *Sphagnum*-dominated bog would be expected to create a self-sustaining ecosystem with low wildfire risk, and thus negate the case for reintroduction of managed burning in future.

3.3.2.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation. Changes in precipitation patterns and temperature extremes is expected to increase the risk of wildfire in the UK, although the greatest increase in risk is anticipated in the South and East of the UK, which does not correspond to the location of shallow peats in the UK (Arnell et al., 2021). There are some concerns that climatic conditions could be less favourable for peat formation in some upland areas in future, but in general natural or restored peatlands will be more resilient to climate change that degraded ones.

3.3.2.9 Climate Factors / Constraints

Peats more at risk from drought may benefit more from management for fire suppression, particularly during the transition from historic burn-management to a more natural, fire-resistant vegetation community. Local climate and vegetation communities may interact with burning regimes to change the effect of reduced burning on carbon sequestration.

3.3.2.10 Benefits and Trade-offs to Farmer/Land manager

Where (and if) reduced burning inhibits rearing of grouse or other financially and culturally valued activities typically associated with fire management on moorland there may be a reluctance towards uptake.
The risk of wildfire would be a significant disincentive due to associated damage to land and risks to safety, but as discussed above the link between burning management and wildfire on peat is complex and may be mitigated.

3.3.2.11 Uptake

There may be economic ramifications from a change in burning regime on activities associated with grouse rearing that provide a barrier to uptake.

3.3.2.12 Other Notes

None

3.3.3 ECCM-030: Restore/manage upland and lowland peatlands including blanket bog and raised bog

Food and fibre production	Area under production or management intensity outside of ELM	T***
Global, regional & local climate regulation	Above ground carbon sequestration	TD*
	Below ground carbon sequestration	***

Duplicate evidence base:
Manage hydrology in wetland habitats to restore functional processes [**ECCM-032**]; enhance/ manage wetland habitats [**EBHE-164EM**]; Enhance/ manage wetland habitat mosaics, including creating the appropriate hydrological conditions [**ECCA-007EM**]; Rewet moorland (including common land), e.g. through appropriate traditional grazing techniques [**EBHE-216**]; Restore peatland vegetation [**ECCM-033**]; Manage the dominance of graminoid or ericaceous species on bog by hydrological restoration, light summer grazing and cutting [**ETPW-158**]; Use controlled grazing (bogs and peatlands) [**ECCM-031**]; **EHAZ-063** Block drains, ditches and grips; **ECCM-030Z**: Restore/manage peat extraction sites; **ECCM-039** Restore areas of farmed peatland to wetland; Create / maintain raised water level areas by appropriate installation and operation of water level controls [**EBHE-212**]; **EHAZ-129EM** enhance or manage fen; **ETPW-155** Remove grazing from recovering peatland, susceptible habitats and sensitive vegetation

Specific summaries are presented below for a subset of specific peatland management and restoration options where a sufficient body of evidence was available to do so, and confidence varied significantly from that of the main assessment.

3.3.3.1 Causality

There is abundant evidence that the successful restoration of peatland can reduce the rate of carbon loss and GHG emissions from peatland, through both empirical study and modelling. The empirical evidence base for specific restoration managements is lacking in some areas, particularly for methods required to achieve successful restoration in a large scale in lowland systems. However, research to address these knowledge gaps is ongoing in the UK.

Peatlands contain substantial reserves of sequestered carbon, which, as a result of degradation and historical drainage, are currently the source of significant green-house gas emissions at the national scale. Although estimating their true carbon content is challenging, a minimum estimate of 3,121 Mt C has been suggested for UK peatlands (Lindsay, 2010). Estimates of the total carbon content stored in peat are uncertain largely due to variation in the depth of peat, and estimates of carbon storage at depth (Gregg et al., 2021). Blanket bog peats over 6m deep in Dartmoor were estimated to contain over 5200 tC ha $^{-1}$ (Fyfe et al., 2014), whilst extrapolations from the CS2007 data have estimated blanket bogs hold 259 tC ha^{-1} in their top 50 cm (Ostle et al., 2009). Peat in raised bogs has been recorded to exceed depths of 12m in some places and hold the largest carbon stores of peatland per unit area (Gregg et al., 2021; Natural England, 2010). Carbon stores in fenland peat soils are less well quantified, with preserved deep peats estimated to store 144 Mt C, which is comparable to the 186 Mt C in the more extensive wasted fenland peats (Natural England, 2010). Following drainage, degradation and conversion to agricultural land use, UK peatland is emitting an estimated 18.8 $MtCO_2$-eq yr^{-1}, as estimated by Brown et al. (2021), which was sufficient to convert the entire UK LULUCF inventory from a GHG sink into a net GHG source. Therefore, the critical value of peatland restoration in the context of carbon sequestration is to minimise the loss of historically sequestered carbon as a result of peatland degradation. Whilst some restored peatland may return to a net carbon sink, this will be a comparatively small component of the overall emissions abatement that could be achieved through peatland restoration.

Restoration of peatland is actively underway across a wide area of the UK, with an estimated 95000 ha of UK peatland having been subject to some form of active restoration between 1990 and 2014, which is thought to have reduced emissions from peatland by 423 kt CO_2e yr^{-1} (Evans et al., 2017). However, most of these projects have been carried out at relatively small spatial scales when compared to the scale of restoration considered in national modelling exercises, with a strong bias towards restoration in modified upland bogs (Element Energy & UKCEH, 2021; Evans et al., 2017; Thomson et al., 2020), and so there is some uncertainty around the levels of success that can be anticipated for peatland restoration at a national scale. Since this assessment was undertaken, the rate and extent of peat restoration across the UK have increased dramatically, particularly in the uplands, with major ongoing investments in restoration via (for example) the Nature for Climate Fund in England, and Peatland Action in Scotland. However, restoration activities in lowland peat, particularly in areas currently managed for agriculture, have been far less extensive to date, and initiatives to mitigate emissions from peatlands remaining under agriculture are still at an early stage; the Defra Lowland Agricultural Peat Task Force will issue recommendations on this topic during 2022.

The UK's peatlands cover an estimated 3 million hectares (12.2% of land area), composed of the following categories (Evans et al., 2017):

- 22% of peats in a near natural condition;
- 41% semi-natural peats, typically subject to drainage, burning and grazing;
- 8% under grassland;
- 7% of peats under arable land use;

- 16 % covered by woodland;
- Industrial and domestic peat extraction occurring over approximate 4600 ha and 145500 ha, respectively.

Although near-natural peats act as a significant net carbon sink for CO_2, this is largely counterbalanced by emissions of CH_4 making their overall GHG impact close to neutral (in CO_2 equivalent units, and based on GWP_{100} values) acting as modest net GHG sinks long term (Evans et al., 2017). Degraded peats act as net sources of GHG emissions (CO_2, CH_4 and N_2O) and the potential impact of restoration per unit area varies with current rates of emissions, estimated for the UK by Evans et al. (2017). There emissions are a result of a combination of decomposing soil carbon, agricultural inputs (particularly for N_2O emissions) and livestock emissions.

The greatest potential dividends come from restoring areas that emit greenhouse gases at the highest rates, which occur in cropland (37.61 $tCO_2e\ ha^{-1}yr^{-1}$), followed by intensive grassland (27.54 $tCO_2e\ ha^{-1}yr^{-1}$), domestic and industrial peat extraction (13.37 and 13.28 $tCO_2e\ ha^{-1}yr^{-1}$), eroding modified bog (13.28 $tCO_2eq\ ha^{-1}yr^{-1}$ if drained, 12.17 $tCO_2eq\ ha^{-1}yr^{-1}$ if undrained), and extensive grassland (13.03 $tCO_2e\ ha^{-1}yr^{-1}$) (Brown et al., 2021). After accounting for the area of drainage-based agriculture on lowland peats, Evans et al. (2017) estimate that this alone accounted for 70% of the emissions from peat. However, competition with food production is likely to make these areas the most challenging to restore. The emission factors for modified fen and bog (low level degradation) are dependent on vegetation cover, level of erosion and drainage (Brown et al., 2021). Although restoration practices and emissions factors vary across re-wetted bog and fen, the overall emissions abatement potential of both habitats following restoration is judged to be similar (Element Energy & UKCEH, 2021). The reduction of emissions rates to those of near-natural fen and bog is dependent on the ultimate success of restoration practices. Some discussion of specific restoration practices and the underlying evidence is provided below.

Managing water tables

Existing evidence shows that the restoration of water tables is a vital component of successful wetland restoration (Evans et al., 2021), and preserving sequestered carbon, although evidence supporting specific restoration practices at scale is less consistent. Raising the water table supresses the break-down of organic compounds, which reduces the production of CO_2. Enzymes that degrade soil organic carbon are impaired by the absence of oxygen and the presence of phenolic compounds, which are preserved under typical wetland conditions (Whitmore et al., 2015). Increased oxygen availability in drained peatlands promotes breakdown by aerobic decomposers, resulting in the release of CO_2 (Alison et al., 2019).

Peatlands have been drained largely through the installation of drainage ditches, in association with agricultural activity, extraction and forestry (Alison et al., 2019). In addition lowering the water table, drainage ditches can be subject to erosion and facilitate increased losses of carbon as DOC and POC, and although the ultimate fate of eroded organic matter is often unclear it is typically viewed as a form of emission. Evans et al. (2017) estimated of an emissions factor for rewetted bog of 0.81 $tCO_2e\ ha^{-1}yr^{-1}$and for rewetted fen of 6.37 $tCO_2eq\ ha^{-1}yr^{-1}$.

The main mechanisms by which the water table is raised are by ditch blocking or larger scale hydrological management as necessitated by specific projects (Evans et al., 2017). In some examples, continued erosion in unrestored surrounding areas due agricultural land use necessitated active maintenance of water levels on the restoration site (The Great Fen Project, 2010).

Recent research by Evans et al. (2021) has shown the potential benefits for net carbon sequestration from raising the water table to near the soils surface. This study was based on long term monitoring of eddy covariance data at 16 sites across the UK and Ireland, using a space for time substitution to understand the effect of water table depth on emissions. The study suggests that by raising water tables until the annual effective water table depth is 10 cm from the surface significant reductions in the rate of GHG emissions were reported achieved, with the net sequestration of least -3 t $CO_2\ ha^{-1}\ yr^{-1}$ for every 10cm the water table was raised until 30cm from the surface. When the water table is less than 10 cm from the surface estimates of the net effect on GHG emissions was associated with greater uncertainty, and the potential for an increase in CH_4 emissions. Raising water levels *above* the peat surface has the potential to generate high rates of CH_4 emission, offsetting the mitigation benefits of reduced CO_2 emissions.

Most temporal studies of the effects of raising water tables on carbon balances are relatively short term, and show mixed results for soil carbon content and losses as DOC from the blocking of drainage ditches (Evans et al., 2018; Turner et al., 2013; Williamson et al., 2017). Holden et al. (2011) reported that the restoration of drained blanket bog was not fully achieved 6 years after drain blockage with hydrological dynamics showing intermediate properties between those of a drained and intact bog. It has been suggested that blanket bogs may be less responsive to rewetting due to low hydraulic conductivity and the effects of subsidence, which may have lowered the surface of the degraded bog to the reduced water table (Williamson et al., 2017). However, Gregg et al. (2021) emphasise that this should not deter restoration efforts in blanket bog systems, and that raising of water tables should be considered a “no-regrets option” as benefits may be realised over longer time periods.

Vegetation restoration

The restoration of peatland vegetation can include scrub removal, a reduction in grazing pressure or burning, and the reintroduction of peatland specialists, including peat-forming vegetation. We consider woodland clearing separately below. The evidence base for the impacts of specific vegetation managements on carbon sequestration and emissions short term is variable between studies. When reviewing records of peatland management published in the peatland compendium (Holden, 2008), Evans et al. (2017) found that few projects which carried out vegetation management alone (without managing the water table) succeeded in transitioning peatland from one state of degradation to a less degraded classification of peatland. However, logic indicates that active management of vegetation will be necessary for successful restoration in cases where natural succession is likely to be slow or uncertain, particularly when stabilising bare peat (Gregg et al., 2021).

Shuttleworth et al. (2015) show that the stabilisation of ditch gullies by re-establishing peatland vegetation can significantly reduce carbon losses as POC by two orders of magnitude in approximately 10 years. A recent study in North-West Germany found that although rewetting raised bog accounted for a 75% reduction in CO_2 emissions compared to intensive grassland, emissions of CH_4 remained high. However, emissions were net negative following subsequent *Sphagnum* reintroduction and top soil removal, to promote nutrient poor and acidic conditions, although the removal of the topsoil results in gross carbon emission (Huth et al., 2021). Grazing intensity is associated with an increase in net GHG emissions from peat, in addition to greater emissions from livestock (Evans et al., 2017). However, a modelling study of the impact of upland grazing on GHG fluxes in peatland suggests that a significant proportion of emissions was produced by the livestock directly rather than from soils or lost NPP (Worrall & Clay, 2012), therefore reducing stocking densities may provide a dual benefit in terms of reducing GHG emissions. A separate study on a long-term experimental site found that areas burnt every 20 years and grazed by sheep had the shallowest water tables along with greater compaction of peat and higher rates of runoff, whilst those which had not been burnt or grazed had deeper water tables and less peat compaction (Clay et al., 2009).

The change in above ground carbon stocks that occur as a result of peatland restoration will be variable, and depend strongly on the starting land use (extensive grazing, intensive grazing, arable). When restoring peat under moorland vegetation, some loss of total above ground carbon stocks may occur, but this change is expected to be small in comparison to the prevention of soil carbon losses each year (Gregg et al., 2021).

3.3.3.2 Co-Benefits and Trade-offs

Near-natural peatlands are also associated with reduced emissions of N_2O when compared to drained and cultivated peatlands. Emissions of CH_4 are typically higher from the surface of natural and re-wetted sites versus drained sites, however CH_4 emissions from drainage ditches can be extremely high, partly balancing out the impacts of peat drainage (Peacock et al., 2021).

Peat restoration can also confer water quality benefits as a result of reduced DOC losses, through reduction of peat erosion. As a result, water treatment companies have commercial incentive to fund peatland restoration to reduce treatment costs (Smyth et al., 2014), although the empirical evidence that DOC loss can be substantially reduced by peat restoration is mixed (Evans et al., 2016).

Rewetting peatlands can also reduce the risk of fire on peatlands and associated carbon losses.

Restored peatlands have the potential to act as water storage (particularly in lowland floodplain areas) and may retard surface flow relative to degraded areas in the uplands, thereby acting as assets towards regulating flood risk. However this may be highly site dependent, and again the empirical evidence is limited. Many cultivated lowland peatlands have undergone several metres of land subsidence, to the extent that large areas are now below sea level, and therefore clearly at risk of flooding if not actively managed. Sites where the water tables are artificially maintained at higher levels may also put pressure on water reserves in the future due to increased water demand, and intensifying drought risk from climate change, although evidence from eddy covariance studies does not suggest that water losses from wetland vegetation are substantially higher than those from grassland or cropland on peat (an assessment of this evidence is ongoing for the Defra Lowland Agricultural Peat Task Force).

Despite their high emissions footprint, agricultural lowland peatlands also provide high quality agricultural land (notably for horticulture) and are therefore subject to significant competition for land area. Large scale restoration of lowland peats could therefore result in diminished capacity for domestic food production and reduced UK food security.

3.3.3.3 Magnitude

Across all estimates, anticipated rates of emission abatement due to peatland restoration are considered high, relative to other activities with the potential for GHG abatement (Element Energy & UKCEH, 2021).

Evans et al. (2017) estimate that the total potential of between -4331 ktCO_2e yr^{-1} could be achieved by 2050 following the cessation of all peat extraction, 25% restoration of degraded lowland peat and 50% restoration across each of England, Scotland, Wales and Northern Ireland.

Thomson et al. (2020) estimated that annual emissions from peatland can be shrunk from +24 Mt CO_2e yr^{-1} to +14 Mt CO_2e yr^{-1} 2050 the UK under their Balanced Net Zero pathway by 2050, assuming the following is achieved: 100% restoration for upland grassland on peat by 2045 (1193 kha); 50% restoration for lowland grassland by 2050 (107kha); 75% restoration of cropland by 2050 (as a mixture of paludiculture and water table management, 142 kha); 100% restoration of peat extraction sites by 2035 (136 kha) and 18% restoration on forest sites (82kha).

Increasing the area of restored peat, and in particular the area of lowland peat under agriculture restored, results in a greater abatement of carbon emissions over time under all scenarios.

Modelling of the potential for greenhouse gas removal in the UK by Element Energy & UKCEH (2021) estimates that maximum technical potentials of -4.7 MtCO_2e yr^{-1} abatement are possible, largely constrained by areas of highly degraded peatland available for restoration. A theoretical abatement rate of -8.7 MtCO_2 e yr^{-1} is reported possible if 100% of modified upland bog was included in scenarios.

It should be noted that projections are sensitive to variable assumptions of the level of success of restoration when rolled out on mass across the full area subject to management, and that scenarios that assumptions of responses to peat restoration under forest are subject to greater uncertainty (Element Energy & UKCEH, 2021; Evans et al., 2017).

3.3.3.4 Timescale

>10 years
A period of 10 years has been assumed sufficient for peatland restoration to occur, after which vegetation NPP is thought to reach values typical for peatland vegetation (Element Energy & UKCEH, 2021). Gregg et al. (2021) state that interventions for peatland restoration take at least 5 years for systems to stabilise, when the objective is a specific vegetation community and multiple decades where the objective is functional restoration.

Active management for carbon formation during peatland restoration has be proposed as a method of delivering higher rates of sequestration faster, but this methodology has not been demonstrated in practice (Element Energy & UKCEH, 2021).

3.3.3.5 Spatial Issues

Restoration actions should be targeted towards to the specific needs of a given site. Requirements for successful restoration are likely to be variable, as are the potential benefits for carbon sequestration.

3.3.3.6 Displacement

Restoration of peatland will entail the reduction of grazing pressure and in some cases food production, in which cases activities may be displaced elsewhere within or outside of the UK without an associated shift in demand. However the area of cultivated peatland is fairly small relative to the total area of UK agricultural land, so an overall negative impact on UK food production is not considered inevitable, although barriers to implementation (economic or otherwise) are likely to be high.

3.3.3.7 Maintenance and Longevity

In some cases continued investment and management with be required to maintain changes in water table, given the context of surrounding landscapes. Changes in water table requiring active long-term water and/or vegetation management may be susceptible to reversal at a later date if adequate means of support are not in place, which may reverse the direction of trends in peatland emissions. However, restoration that leads to hydrologically and ecologically self-regulating systems could be considered highly durable (Element Energy & UKCEH, 2021).

3.3.3.8 Climate Adaptation or Mitigation

Restoring peatland is likely to contribute substantially to climate change mitigation in the UK.

3.3.3.9 Climate Factors / Constraints

Increasing weather extremes due to climate change may put increasing pressures on rewetted areas as sites of flood mitigation and potentially with conflicting demands for water use during drought. Some climate envelope models have suggested that peatlands will become inviable in some climatically marginal areas of the UK with concerning implications for the fates of carbon currently stored there (Mulholland et al., 2020b)

3.3.3.10 Benefits and Trade-offs to Farmer/Land manager

Economic barriers to uptake are likely to vary significantly across upland and lowland peats, with lowland areas that are more commonly used for intensive grazing or cropland having significant financial disincentives for restoration. Element Energy & UKCEH (2021) estimate that the capital costs of restoring peatland is approximately £26-48 per tCO_2 removed, although these figures should be considered highly uncertain for lowland areas in particular, given that few large-scale restoration projects have been attempted to date.

3.3.3.11 Uptake

The existence of the Peatland Code (https://www.iucn-uk-peatlandprogramme.org/) provides an established framework by which peatland restoration could be approached, potentially facilitating uptake of successful management programs.

The restoration of peatland is included in the commitments of the England Peat Action Plan, and is supported by the Nature for Climate Fund, which aims to fund the restoration and future maintenance of 35,000 ha of peatland by 2025 (DEFRA, 2021a). Similar schemes are in operation in the other UK countries.

3.3.3.12 Other Notes

The evidence base underlying certain specifics of peatland restoration would benefit from further research and investment. Areas of particular concern highlighted in the literature are the losses of carbon associated with POC loss from erosion, burning and CO_2 measurements from wasted peatland under cropland and intensive grassland (Evans et al., 2017). In addition there is high uncertainty associated with the restoration of peatland currently under conifer plantations; this is discussed more below.

3.3.4 ECCM-034: Remove non-peat habitat vegetation

3.3.4.1 Causality

Food and fibre production	Area under production or management intensity outside of ELM	T**
Global, regional & local climate regulation	Above ground carbon sequestration	**
	Below ground carbon sequestration	**D*** **

Afforested peat is associated with high rates organic matter loss as CO_2 as a result of drainage. Estimated rates of carbon loss from peat under forest are associated with a high degree of uncertainty however, due to difficulties distinguishing between fluxes due to forest respiration and litter decomposition, and the breakdown of peat. It was estimated that peat under forest has an emissions factor of between 1.15 and 5.46 tCO_2-eq ha^{-1} yr^{-1} in 2019 (Brown et al., 2021), but this figure does not take into consideration the sequestration of CO_2 by woodland growing on peat, or the long-term fate of that CO_2 post-harvest of the woodland.

In the wider literature, the net impact of forest removal on the GHG balance is unresolved (Artz et al., 2013; Hambley et al., 2018). In the early stages of restoration it is likely that CO2 emissions will increase, e.g. due to decomposition of brash, but over long periods (> 10 years), as hydrology and vegetation recover, the (limited) evidence suggests that forest-to-bog restoration will lead to the re-establishment of net CO_2 sinks. It is widely accepted that the planting of forests on peatland is associated with significant carbon emissions, compared to planting forests elsewhere, as a result of physical disturbance and draining during the initial planting and through subsequent transpiration (Gregg et al., 2021). The longevity of carbon stored in peat also far exceeds the longevity of carbon in wood, suggesting that restoration of peatland would prove beneficial in the long term (e.g. centuries) (Evans et al., 2017; Gregg et al., 2021).

Where the non-peat habitat vegetation is scrub and heath, the net benefits of restoration are better established, if removal is part of a wider restoration program (Evans et al., 2017).

For a more general discussion of the evidence underpinning vegetation restoration, see **ECCM-030**.

3.3.4.2 Co-Benefits and Trade-offs

See Trade-offs/Co-benefits evidenced under **ECCM-030.**

3.3.4.3 Magnitude

The emissions factor for peat under forest is estimated at between 1.14 and 5.46 tCO_2-eq $ha^{-1}yr^{-1}$ (Brown et al., 2021). This is lower than the estimate provided by Evans et al. (2017) of 9.91 tCO_2-eq ha^{1} yr^{-1},as a result of an average increase in forest age. The estimated emissions from rewetted bog and fen are 3.91 and 8.05 tCO_2-eq $ha^{-1}yr^{-1}$, respectively, due to high rates of CH_4 production (Brown et al., 2021). However, emissions from rewetted semi-natural bog are estimated as equal to those in near-natural bog at -0.02 tCO_2-eq $ha^{-1}yr^{-1}$ (Brown et al., 2021). Estimates of emissions from rewetted peatland are largely not based on previously afforested systems however, and do not include the effect of restoration on of forest carbon sequestration stocks and sequestration, in the context of the forestry lifecycle.

The overall magnitude of benefits, including forest carbon losses, are not well established.

For comparison, estimates for production forestry in Wales (not limited to peat) suggest that, including cross sectorial emissions, the mean climate change mitigation potential between 2020 and 2050 is on average -3. 5 tCO2e $ha^{-1}yr^{-1}$, however values range between -14.5 tCO2e $ha^{-1}yr^{-1}$ and +2.65 tCO2e $ha^{-1}yr^{-1}$ (Matthews, 2020).

The maximum potential carbon uptake from woodland creation in the UK is site dependent. Matthews et al. (in prep.) presents an analysis of carbon sequestration and storage over time as a result of afforestation in nine study sites with varied compositions and managements. The estimated rate of carbon sequestration in living woody biomass during a 100 year period from the time of the creation of these example woodlands, was in the range of -1 to -3 tC ha^{1} y^{1} (-3.7 to -11 tCO_2 ha^{1} y^{1}), commonly taking a value of -1.4 tC ha^{1} y^{1} (-5 tCO_2 ha^{1} y^{1}). This summary excluded stands managed for clearfelling.

Matthews et al. (in prep.) also report model estimates for 11 contrasting options for woodland creation involving tree species mixtures and management practices relevant for the UK. These options include natural recolonisation of non-wooded land with broadleaf trees and 'light' subsequent management, examples of mainly moderate- and fast-growing commercial coniferous woodlands, and managed woodlands composed of complex tree species mixtures. The mitigation contributed by the woodland options contrasts in magnitude over time, with different options providing the most mitigation benefits at different times (between 2022 and 2100 and beyond) and in different ways (involving direct carbon sequestration or GHG emissions savings through provision of wood products, to varying extents). It should be noted that the different woodland options are not always interchangeable on the same site or in the same location within the UK. Planting 1 hectare of woodland in 2022 is estimated to result in net carbon sequestration in woodlands and wood products over the period from 2022 to 2050 of between -0.8 and -13.8 tCO_2 ha^{-1} y^{-1}, with a mean estimate for all 11 options of -5.7 tCO_2 ha^{-1} y^{-1}. If GHG emissions savings arising from utilisation of additional wood products and wood fuel are allowed for, these estimates increase to between -1.7 to -32.0 tCO_2 ha^{-1} y^{-1}, with a mean estimate for the 11 options of -12.4 tCO_2 ha^{-1} y^{-1}. Matthews et al. also report estimates of mitigation per hectare for a programme of woodland creation based on the 11 example woodland options over 25 year period starting in 2022. Such a programme is estimated to provide net carbon sequestration of between -0.16 and -4.9 tCO_2 ha^{-1} y^{-1}, with a mean estimate for all 11 woodland options of -1.6 tCO_2 ha^{-1} y^{-1}. (Hence, for example, a programme to create 20,000 ha of woodland over 25 years would provide carbon sequestration of between -3 and -98 $ktCO_2$ y^{-1}, with a mean estimate of -32 $ktCO_2$ y^{-1} for a combination of the woodland options). If GHG emissions savings arising from utilisation of additional wood products and wood fuel are allowed for, the per-hectare estimates for the 25 year programme are -0.12 to -10.7 tCO_2 ha^{-1} y^{-1}, with a mean estimate for all 11 options of -3.3 tCO_2 ha^{-1} y^{-1}.

3.3.4.4 Timescale

>10 years.

Although the removal of vegetation can be argued to be immediate, studies of peatland restoration show that restoring near-natural carbon fluxes can take many years.

3.3.4.5 Spatial Issues

No assessment.

3.3.4.6 Displacement

Removal of forestry may displace activities onto other land uses. However, assuming this activity is displaced out of peatland habitats the net impacts of this are likely to be beneficial for carbon fixation.

3.3.4.7 Maintenance and Longevity

For removal to be effective it should be supported by more general restoration of peatland which can is generally carried out over many years, and may continue indefinitely. The removal on non-native vegetation may require ongoing management, due to the potential for the re-establishment of non-native species (Anderson & Peace, 2017).

3.3.4.8 Climate Adaptation or Mitigation

Will create a more sustainable wetland system.

3.3.4.9 Climate Factors / Constraints

Focus on wetlands which may be more sustainable in the longer term, when subjected to increased frequency of droughts.

3.3.4.10 Benefits and Trade-offs to Farmer/Land manager

Removal of non-native vegetation may financially disadvantage land managers due to the loss of revenue from forestry or associated practices, such as grouse rearing.

3.3.4.11 Uptake

Peatland restoration is already underway in the UK on a notable scale. The Peatland Code is also likely to facilitate uptake of restoration and private funding to support activities.

3.3.4.12 Other Notes

None

3.3.5 EHAZ-137: Use paludiculture

3.3.5.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T*** **
Global, regional & local climate regulation	Above ground carbon sequestration	**N**
	Below ground carbon sequestration	**L*** **

Paludiculture is the practice generating commercial crops (food or fibre) in wet agriculture or forestry on peatlands, and may first involve the rewetting of soil. Trials of paludiculture in UK lowland peatlands are currently underway, to determine the potential of this practice for preserving sequestered carbon in peatlands and reducing emissions associated with current lowland farming practices on peat. As such, evidence of the overall impact of paludiculture on carbon sequestration and emissions abatement in the UK are currently unknown.

Potential applications of paludiculture in the UK include the commercial farming of Sphagnum and biofuel production, wetland crop production (including potentially celery and watercress) the production of fodder and fisheries applications (Mulholland et al., 2020b). The overall impact of the industry on carbon sequestration will be dependent on wider components of the agricultural lifecycle, and its effect on the risk of agricultural displacement. A risk of displacement of food production in lowland agriculture is due to a conversion to fibre production (Mulholland et al., 2020b).

The study of paludiculture's GHG footprint is a relatively new field, particularly in the UK, and as such there is insufficient evidence to robustly estimate the impact of paludiculture on carbon sequestration and GHG emissions more generally. Studies conducted thus far all indicate that paludiculture will have a smaller carbon footprint that the intensive lowland agriculture it is likely to replace spatially, with some indicating that sites act a net carbon sinks when water tables are highest (Mulholland et al., 2020b). However these assessments do not take into account the off-site carbon emissions associated with these farming systems and so a full evaluation is not possible at this stage. There is strong evidence that raising average annual water table within peatlands can significantly reduce greenhouse gas emissions from peat, and this would apply to paludiculture projects in the same way as for conventional restoration (Element Energy & UKCEH, 2021; Evans et al., 2021; Evans et al., 2017).

3.3.5.2 Co-Benefits and Trade-offs

Limited biodiversity benefit. Flood risk management. Reduced risk of peat fires. Reduced soil rates of peat soil erosion from wind.

High water tables have a risk of resulting in elevated methane production which could offset some of the benefits of reduced carbon dioxide emissions (Evans et al., 2021).

Because paludiculture crops are able to withstand flooding, it is possible that areas of land managed for paludiculture could also provide winter flood storage, comparable to the 'washlands' which have formed an intrinsic part of the water management of areas such as the Fens since they were first subject to large-scale drainage in the 17th century. This could provide a significant co-benefit by providing flood protection for adjacent urban and agricultural areas, as well as providing a water store that could help to alleviate water scarcity during summer (Mulholland et al., 2020). The relative benefits of this water storage and release potential, versus the potential disbenefit of higher summer water demand of paludiculture crops, is an area of uncertainty.

3.3.5.3 Magnitude

Estimates of the net sequestration potential for paludiculture in the UK are unknown, although trails are currently underway. The impact of paludiculture could be lower than that of peatland restoration (see section 3.3.3) because it typically involves removal of biomass that might otherwise have contributed to peat formation. On the other hand, the more controlled water management and vegetation under paludiculture could offer potential for optimisation of CO_2 sequestration and avoidance of CH_4 emissions, and some C sequestration could also occur via the incorporation of harvested wetland products into long-lived building materials such as fibreboard (Mulholland et al. 2020).

3.3.5.4 Timescale

Unknown – whilst the successful raising of the water table should result in a rapid reduction in emissions from peatland, the difficulties of raising and maintaining water tables at large spatial scales and complex agricultural landscapes are unknown.

3.3.5.5 Spatial Issues

Targeting is crucial
Collaboration is beneficial
Paludiculture will have the largest incentives where conventional agriculture is no longer viable, or is likely to become inviable in the near future. However, there are limits on the spatial scale over which paludiculture can be viable, meaning that where areas of wasted peat are highly fragmented, incorporating mineral soils into paludiculture may be necessary (Mulholland et al., 2020). The sustainability of paludiculture will vary with the requirements of restoring the water table sufficiently to produce wetland conditions (Element Energy & UKCEH, 2021; Mulholland et al., 2020)

3.3.5.6 Displacement

The risk of displacing lowland agricultural activity is dependent on the capacity of paludiculture to support equivalent food production. If equivalent production is not achieved, it stands to reason that agricultural activities will be displaces either nationally or internationally, with potential implications for domestic food security. However, for sites that are approaching the end of their agricultural viability, this transition may also occur in the absence of paludiculture.

3.3.5.7 Maintenance and Longevity

Active management of the water table is highly likely to be an ongoing requirement in many lowland systems, if not to maintain the water table at sufficient levels for paludiculture, then to manage regional flood and drought risk (Mulholland et al., 2020).

3.3.5.8 Climate Adaptation or Mitigation

Paludiculture has the potential to act as both climate change mitigation, through the avoidance of GHG emissions typically associated with lowland agriculture, and climate change adaptation, as changing climatic conditions make traditional farming methods less viable. However requirements for additional water could present long-term challenges in water-scarce areas (see below)

3.3.5.9 Climate Factors / Constraints

Paludiculture is reliant on the restoration of water tables to sufficient levels. In some areas of peat-based lowland agriculture in the UK, this may be challenging to maintain throughout the year. Balancing the demands on water supplies from paludiculture with those of the surrounding area, including managing flood and drought risk and supporting continued crop production, could prove to be a constraint in some areas. The relative water demand of paludiculture relative to conventional agriculture on peat remains an area of uncertainty.

3.3.5.10 Benefits and Trade-offs to Farmer/L-and manager

The potential benefits and trade-offs for land managers under paludiculture are difficult to fully anticipate at this stage in the practices development. Viability of paludiculture as a business is likely dependent on the development of machinery and skills currently not present in the industry and the development of suitable markets and supply chains. The development of carbon markets could however offer financial incentives for a transition to paludiculture.

3.3.5.11 Uptake

The transition to a raised or restored water table in what is currently lowland agricultural land presents a significant barrier to uptake of paludiculture, due to conflicting requirements for water management at present. Sourcing required volumes of water, particularly in summer, may also be a barrier to uptake in the East of the UK, owing high rates of evapotranspiration expected from paludiculture systems (Mulholland et al., 2020). Management of water at the multiple farm scale by Internal Drainage Boards also makes it difficult for many individual farmers to change their water management without impacting on neighbouring farms.

More generally, the UK agricultural sector is likely to require specialist agricultural machinery and skills for paludiculture to become viable (Johnson et al., 2017).

3.3.5.12 Other Notes

None

3.3.6 ECCM-038: Raise water levels in areas of farmed peatland and adapt farming systems accordingly

3.3.6.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	T**
Global, regional & local climate regulation	Above ground carbon sequestration	N
	Below ground carbon sequestration	***

Recent research by Evans et al. (2021) has shown the potential benefits for net carbon sequestration from raising the water table to near the soils surface. This study was based on long term monitoring of eddy covariance data at 16 sites across the UK and Ireland, using a space for time substitution to understand the effect of water table depth on emissions. The study suggests that by raising water tables until the annual effective water table depth is 10 cm from the surface significant reductions in the rate of GHG emissions were reported achieved, with the net sequestration of least -3 t CO_2 ha^{-1} yr^{-1} for every 10cm the water table was raised until 30cm from the surface. When the water table is less than 10 cm from the surface estimates of the net effect on GHG emissions was associated with a large increase in uncertainty, and the potential for an increase in CH_4. These results are supported by the wider literature for the effectiveness of peatland restoration and water table management (Evans et al., 2017). This management change has the potential to reduce the loss of peat carbon stores without necessitating a wider change in farming system, as would be required with paludiculture (see section 3.3.5), although adaptation will be required, likely involving the cultivation of strains with a tolerance to higher water tables and the challenges of large scale water table management (Evans et al., 2021).

3.3.6.2 Co-Benefits and Trade-offs

Limited biodiversity benefit. Flood risk management. Reduced risk of peat fires. Reduced soil rates of peat soil erosion from wind.

3.3.6.3 Magnitude

By raising water tables until the annual effective water table depth is 10 cm from the surface significant reductions in the rate of GHG emissions were reported achieved, with the net sequestration of least -3 t CO_2 ha^{-1} yr^{-1} for every 10cm the water table was raised until 30cm from the surface.

3.3.6.4 Timescale

Unknown – whilst the successful raising of the water table should result in a rapid reduction in emissions from peatland, the difficulties of raising and maintaining water tables at large spatial scales and complex agricultural landscapes are unknown.

3.3.6.5 Spatial Issues

Targeting is crucial. Collaboration is beneficial.

Increasing water tables will have the largest impact on emissions in agricultural systems where existing emissions are largest. However, there are practical limits on the spatial scales over which raising the water table can be viable, and where peats are highly fragmented, incorporating mineral soils may be necessary (Mulholland et al., 2020).

Following expert opinion, some crops on peat are expected to be able to accommodate higher water tables during the growing season without negatively impacting growth, whereas other crops will not be able to, and this will either cause displacement, or mean that water tables should ideally only be raised outside of growing season. These nuances will be extremely important in any practical application.

3.3.6.6 Displacement

The risks of displacement from this action are largely unknown, and will be heavily dependent on the specific practices implemented. Higher water tables during some or all of the year could lead to crop displacement, but expert opinion also suggests it would be possible to avoid or minimise this. Ultimately, it is not possible to reliably estimate the scale of displacement for this action at a national scale, with the evidence currently available.

3.3.6.7 Maintenance and Longevity

Active management of the water table is highly likely to be an ongoing requirement in many lowland systems, if not to maintain the water table at sufficient levels for paludiculture, then to manage regional flood and drought risk (Mulholland et al., 2020).

3.3.6.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.3.6.9 Climate Factors / Constraints

This action is reliant on the restoration of water tables to sufficient levels. In some areas of peat-based lowland agriculture in the UK, this may be challenging to maintain throughout the year. Balancing the demands on water supplies from paludiculture with those of the surrounding area, including managing flood and drought risk and supporting continued crop production, could prove to be a constraint in some areas. The relative water demand of paludiculture relative to conventional agriculture on peat remains an area of uncertainty.

3.3.6.10 Benefits and Trade-offs to Farmer/L-and manager

The development of carbon markets could however offer financial incentives for this action.

3.3.6.11 Uptake

The transition to a raised or restored water table in what is currently lowland agricultural land presents a significant barrier to uptake, due to conflicting requirements for water management at present. Sourcing required volumes of water, particularly in summer, may also be a barrier to uptake in the East of the UK, owing high rates of evapotranspiration expected from paludiculture systems (Mulholland et al., 2020). Management of water at the multiple farm scale by Internal Drainage Boards also makes it difficult for many individual farmers to change their water management without impacting on neighbouring farms. More generally, the UK agricultural sector is likely to require specialist agricultural machinery and skills for paludiculture to become viable (Johnson et al., 2017).

3.3.6.12 Other Notes

None

3.3.7 ETPW-153 Stabilise eroding peat through targeted restoration work

3.3.7.1 Causality

Global, regional & local climate regulation	**Above ground carbon sequestration**	*
	Below ground carbon sequestration	**TD*** **

The stabilisation of eroding peat is considered a “no regrets” action by experts, as peat soils contain large carbon stocks and the stabilisation of degraded peat is typically a prerequisite for further restoration works, including the re-establishment of vegetation cover. The restoration activity itself, potentially involving the use of large mechanised vehicles and short term soil disturbance, is likely to involve net emissions in the short term, although these fluxes have not been quantified. However, the benefits of peatland restoration are likely to offset these emissions after a few years, based on expert opinion.

3.3.7.2 Co-Benefits and Trade-offs

Limited biodiversity benefit.
Flood risk management.

3.3.7.3 Magnitude

No estimates of the impact of peatland stabilisation and associated emissions due to could be found. However, for estimates of the impact of peatland restoration more broadly see section 3.3.3.

3.3.7.4 Timescale

>10 years
Successful peatland restoration is most probably the desired ultimate outcome of targeted stabilisation work, and will be associated with the greatest benefits for net carbon sequestration. Restoration has been estimated to occur over a minimum of 5 years, but multiple decades may be required for full functional restoration.

3.3.7.5 Spatial Issues

No assessment.

3.3.7.6 Displacement

No assessment.

3.3.7.7 Maintenance and Longevity

No assessment.

3.3.7.8 Climate Adaptation or Mitigation

The development of carbon markets could however offer financial incentives for a transition to paludiculture.

3.3.7.9 Climate Factors / Constraints

No assessment.

3.3.7.10 Benefits and Trade-offs to Farmer/L-and manager

No assessment.

3.3.7.11 Uptake

No assessment.

3.3.7.12 Other Notes

None

3.4 Climate Measures – Climate change mitigation and adaption

3.4.1 ECCA-005: Undertake a climate change vulnerability assessment

Only assessed for co-benefits or trade-offs.

3.4.1.2 Co-Benefits and Trade-offs

Global, regional & local climate regulation	**Above ground carbon sequestration**	T
	Below ground carbon sequestration	**T**

The application of climate change vulnerability assessments (CCVA) for enhancement and protection of global climate regulation would necessarily be highly context specific, and there is no empirical evidence base to reliably assess their potential impact of carbon sequestration or climate regulation. However, there is a strong logic chain suggesting that taking account of climate change in habitat management and creation is critical to the long term stability of habitats and their ability to sequester carbon long term.

CCVAs are typically produced by scientists trained in the relevant discipline, ideally in liaison with stake holders and land managers and many are peer reviewed prior to implementation (Timberlake & Schultz, 2019). These assessments provide a framework for examining the impacts of climate change on a range of outcomes, but the diversity of disciplines involved can make the production of a comprehensive and robust assessment challenging (Timberlake & Schultz, 2019).

In principle, anticipating emergent risks due to climate change such as will fire risk or introduced pathogens may enable steps to be taken that would prevent or minimise damage to existing carbon stocks and future sequestration potential. However, appropriate management could be carried out or encouraged outside of the framework of a CCVA, which may be expensive and time consuming to develop (Guidi et al., 2018). An examination of 80 case studies of CCVAs in forest ecosystems at local and national scales found that CCVAs are likely to be most powerful when combined with ongoing funding, abundant supporting data and technical capacity are made available (Guidi et al., 2018). The selection of suitable assessment methods and tools by supporting policy makers is also key (Guidi et al., 2018).

3.4.2 ECCA-026: Plant a range of native species, including trees grown from locally adapted and genetically diverse seed sources, and from more southerly provenances

3.4.2.1 Causality

Food and fibre production	**Area under production or management intensity outside of ELM**	T*
Global, regional & local climate regulation	**Above ground carbon sequestration**	T**
	Below ground carbon sequestration	**LTD****

There is strong evidence that planting new trees can sequester substantial carbon over time, however the contributions of species mixes, seed source diversity and climatic adaptation have varying support. Whilst there are strong logic chains to support the need to select trees that are likely to do well under future climate, empirical evidence for the success of these methods is lacking.

There is very good evidence that planting woodland and trees can lead to substantial carbon sequestration. The rate of sequestration, the size of standing carbon stocks and the longevity of those stocks is highly dependent on the variety of tree planted, local conditions, prior land use, and management of the site (Matthews, 2020). In rare cases, when trees are particularly slow growing, the management and harvesting of trees can emit more CO_2 that the trees themselves have sequestered, and so long-term management of any new planting is critical (Matthews, 2020) . A more detailed discussion of the impacts of planting trees on carbon sequestration for below and above ground carbon can be found at sections on 'Habitat Creation – Woodland' and on 'Woody Features' (3.12 and 3.13).

There is some evidence that multi-species stands in Europe are have larger carbon stocks per unit area, and can promote individual tree growth as a result of more efficient use of space by tree canopies, complementary resource use and potentially facultative interactions (Chamagne et al., 2016; Jucker et al., 2015; Kelty, 2006). In mixes with larger numbers of species, interactions affecting productivity become harder to anticipate and there is significantly less evidence on net effects for carbon sequestration, in the absence of differences in management (Matthews, 2020). The consensus in UK forestry is that maximising carbon sequestration potential requires targeting of the right trees in the right place (Matthews et al., in prep.), which may in some cases include mixtures. Selection of woodland stock can also involve a trade-off between long term potential stocks and the rate at which those stocks might be achieved (Matthews et al., in prep.). The long term management of these stands may have a more significant impact on long term carbon sequestration than species mixes, particularly if comparing woodland in production to non-productive woodland (Lewis et al., 2019; Matthews, 2020).

In addition to affecting the potential for carbon sequestration and stock, there is some evidence that greater species richness can confer greater protection against generalist pest and disease outbreaks (Haas et al., 2011), although the mechanisms and reliability of the diversity 'dilution effect' on pest and disease risk remain poorly understood (Kelty, 2006). New evidence suggests that reduced severity of outbreaks is due to a reduced probability of exposure for highly susceptible individuals, and not due to a generalised reduction of risk of developing disease after exposure (Rosenthal et al., 2021). Rosenthal et al. (2021) also found that higher stand diversity increased the occurrence of symptomatic *Phytophthora ramorum* infections in Bay Laurel in their study site in the US. In non-forest systems, inter-specific interactions between two species grown in mixture has been shown worsen, lessen, or have no effect on the impacts of pest outbreaks (Kelty, 2006). It has been estimated that tree disease in the US is correlated with a reduction of above ground biomass accumulation by 9.33 Tg C yr^{-1} due to insect outbreak and 3.49 Tg yr^{-1} due to disease, compared to trees not recorded as disturbed, controlling for variation in eco-province and carbon prior to any disturbance (Quirion et al., 2021). However the impact in the UK has not been quantified and the degree to which high diversity of stands could reduce that impact is unknown.

Preferentially selecting more southern stock is often referred to as "assisted gene flow" and is based on the principle that more southern populations will be adapted to climatic conditions that are expected to become more prevalent due to climate change as a result of differing phenotypic plasticity and genetic adaptation (Bussotti et al., 2015; Isabel et al., 2020). This practice would introduce traits for drought tolerance to currently susceptible populations faster than could naturally occur (Schueler et al., 2021). Supporting evidence for this practice primarily consist of studies of spatial variation of tolerance to and recovery from drought across population (Aitken & Bemmels, 2016; Conte et al., 2019; Gazol et al., 2018). However, correlational studies cannot demonstrate that tolerance would be preserved following the translocation of populations. Other research has shown that the tolerance and recovery of Scots pine to drought mediated stress is mediated by the favourability of conditions at sites over longer periods of time, not by adaptation over a latitudinal gradient (Bose et al., 2020). An experiment which trialled assisted gene flow, with the goal of elevating NPP in the target population of *Picea mariana* in Canada, found that benefits were limited to the first 15 years after the introduction, after which no effect was detected (Girardin et al., 2021). This suggests that traits expected to be adaptive in the future may not be maintained in the absence of a selective pressure. If these traits are due to phenotypic plasticity, it is possible that they could be recovered. The opinion that some form of assisted gene transfer may be required to allow populations to keep pace with climate change is widely held in the scientific community (Aitken & Bemmels, 2016; Bussotti et al., 2015; Conte et al., 2019; Isabel et al., 2020; Schueler et al., 2021), and the carbon losses that could result from widespread woodland losses due to climate change could be extreme (Ciais et al., 2005; Matthews, 2020). In contrast, locally sources trees may be suited to sequester carbon under current climatic conditions, but may have limited capacity for adaptation (Fritsche et al., 2020).

Overall, there is little empirical evidence to quantify the effect of the practices discussed above on carbon sequestration at this stage, despite a logic chains that suggests their importance for the future preservation of woodland carbon stocks. If these practices are incorporated into woodland management following a climate change risk assessment or woodland carbon plan (see sections 3.12.1 and 4.7.3), the likelihood of a successful outcome would presumably be raised. Significant evidence gaps remain around the potential for populations to incorporate new genetic diversity and whether this would result in an overall increase in carbon sequestration potential compared to other strategies.

3.4.2.2 Co-Benefits and Trade-offs

No assessment.

3.4.2.3 Magnitude

The carbon sequestration potential from afforestation in the UK has been estimated at 11.4 tCO_2 $ha^{-1}yr^{-1}$ under the CCC Balanced Net Zero Pathway scenario (Thomson et al., 2020) with maximum potential of 26.5 $MtCO_2$ by 2050 (Element Energy & UKCEH, 2021).

The impacts of stand diversification and adaptation on carbon sequestration and stocks will be management dependent and vary with species composition and the specific site in question. Reliable estimates of the magnitude of this effect in the UK are not available.

3.4.2.4 Timescale

>5 years
Standing carbon stocks in newly planted trees can take decades to develop. Any resilience provided by greater genetic diversity and adaptation to more southerly climates will likely become more important as climate change progresses. The initial impact of planting trees is often net loss of carbon due to emissions from soil disturbance.

3.4.2.5 Spatial Issues

Targeting is critical on hotspots or equivalent
The site of tree planting (considering soil type, local climate and prior land uses in particular) will be critical to the net carbon sequestration associated with this action. Targeting donor populations which will be able to successfully establish in the UK will be critical for long term success, as will be the combinations of native species used if grown in mixture.

3.4.2.6 Displacement

Planting new trees would displace activities previously carried out on that land including their carbon footprints, unless they are integrated as part of the existing system (for example, silvo-pasture). This displacement could occur internationally or within the UK.

3.4.2.7 Maintenance and Longevity

The management of newly planted trees will have a significant impact on carbon stores (see woodland management bundle, section 3.24). The viability of trees imported from more southern or remote sources (to increase genetic diversity) will also be unknown at the time of planting. Care should be taken when selecting stock to maximise the chance of success, and multiple introductions of new genetic stock may be required.

3.4.2.8 Climate Adaptation or Mitigation

This action has the potential to contribute to both adaptation and mitigation. However, populations may be vulnerable to climate change, particularly to extreme events.

3.4.2.9 Climate Factors / Constraints

Changes in climate will not necessarily replicate conditions historically found in more southern climates, particularly due to increasing climatic extremes. Extreme winter conditions may make the translocation of seed stocks adapted to warmer conditions challenging.

3.4.2.10 Benefits and Trade-offs to Farmer/Land-manager

Managing stands with multiple species for harvesting may introduce logistic challenges, but could also facilitate a diversification of the products produced (Kelty, 2006).

3.4.2.11 Uptake

Administrative burden if importing stock internationally may provide a barrier to entry. Specialist advice may be required to identify suitable stock for the site in questions.

3.4.2.12 Other Notes

Although this action stipulates that native species from more southern ranges, spatially isolated populations of the same species can differ significantly in growth form and dynamics. Selecting species that will grow better under future conditions may impair their growth now.

3.4.3 ECCM-074: Plant, enhance or manage bioenergy crops (e.g. short rotation coppice)

3.4.3.1 Causality

Energy use/ use of renewables		**
Food and fibre production	Area under production or yield and outside of ELM	*
Global, regional & local climate regulation	Above ground carbon sequestration	LT*
	Below ground carbon sequestration	**LTD****

There is evidence that bio-energy crops can provide abatement for CO_2 equivalent emissions if used in place of fossil fuels for energy and heating purposes. However, there is significant uncertainty about whether bio-energy crops can act as a carbon sink, the role of soil loss, N_2O emissions, the long-term sustainability of production, and the impacts of agricultural displacement. These issues warrant further research prior to widespread implementation. A potential limiting factor in the UK will be the suitable area available for cultivation, given competition for food production (Element Energy & UKCEH, 2021).

Bioenergy crops are typically either herbaceous grasses such as *Miscanthus*, which are harvested annually, or short-rotation woody crops, which are harvested less frequently (Lemus & Lal, 2005). The main benefits of cultivating bio-energy crops are in the offset emissions of fossil fuels when harvested biomass is used for power and heating, and in potentially higher rates of carbon sequestration and storage below ground than would be achieve through conventional agriculture (Lemus & Lal, 2005).

For all bioenergy crops, the majority of above ground carbon sequestered in a rotation will be removed for conversion to fuel and ultimately returned to the atmosphere. The utilisation of carbon capture and storage technology may potentially reduce the amount of carbon lost during combustion, but this technology will not be discussed in depth here (see Element Energy & UKCEH, 2021, for more information on the potential of BECCS). Perennial bioenergy crops do have the potential to store significant carbon below ground, to a greater extent than annual crops (Lemus & Lal, 2005). Perennial crops are typically associated with more elaborate root networks and reduced frequencies of soil disturbance than annual crops. However, below ground biomass may also be largely lost when crops are ultimately replanted.

The initial establishment of bio-energy crops has been linked to significant losses of soil carbon, which can delay the achievement of net carbon sequestration in bio-energy crop systems, at the field level. A review of bioenergy crops in the UK by Prosser et al. (2022) concluded that initial losses from soil carbon, during the establishment of *Miscanthus* would be compensated for by the 8th year of growth in a 15-20 year crop, however this balance does not account for the wider lifecycle of bio-energy crops. Harris et al. (2017) compared the impacts of low-input grassland and short rotation coppice on carbon balance and also found evidence of significant losses of soil carbon from the top 30cm of soil under short rotation coppice in the first two years (approximately 20 t C ha^{-1} yr^{-1}) that were not lost in the grassland system. There is also evidence to suggest that rates of soil loss are significantly worse when converting grassland to bio-energy crops, compared to arable (Don et al., 2012; Richards et al., 2017; Whitaker et al., 2018) and Whitaker et al. (2018) suggest targeting degraded arable soils for conversion, to minimise potential losses during land use change, maximise potential gains due to soil restoration, and minimise conflict with food production. Overall, Whitaker et al. (2018) conclude that there is good evidence that low-input bio-energy crops can provide significant CO_2 emissions abatement when substituting for the burning of fossil fuels, provided reasonable yields are attained, low carbon soils are targeted to minimise initial losses, and tensions with food production are minimised. This includes avoiding growing bio-energy crops on previously non-agricultural land, which would typically be associated with substantial carbon losses due to the clearing of vegetation and soil erosion.

3.4.3.2 Co-Benefits and Trade-offs

Transitions from annual crops to perennial bio-energy crops, particularly deep rooting ones, has the potential to influence local hydrology by consuming soil water. Reduced water tables and nutrient inputs could also result in an associated reduction in leaching rates (Whitaker et al., 2018). The establishment of bioenergy crops on wasted peat should be treated with caution, due to the effects of a reduced water table on CO_2-eq emissions from peat.

There is a strong likelihood that the expansion of bioenergy crops will displace food production either domestically or internationally which may result in the expansion of agricultural land elsewhere, negating some of the emissions abatement associate with conversion to bio-energy crops.

The potential for bioenergy crops to emit significant quantities of N_2O has also been subject to debate. The highest rates of N_2O emission have been found bio-energy crops are grown on grassland, or when high rates of fertiliser application are continued, however the effects of N_2O emissions relative to that of soil loss during land use change were reported to be small (Whitaker et al., 2018). Harris et al. (2017) found emissions of N_2O and CH_4 to be negligible under short rotation willow coppice, however this was established on low – input grassland. Despite this, there remains a high degree of variability in reported rates of N_2O emissions from bioenergy crops that warrants further research.

[TOCB Report-3-5D Systems **ECCM-074**] A range of crops can be grown for bioenergy processing in various ways, including short-rotation willow coppice, Miscanthus miscanthus x giganteus and repurposed conventional crops such as oilseed rape or maize. This review considers just novel crops grown for this purpose. Crops such as reed Phragmites australis are considered under **EHAZ-137** Use paludiculture. There would also be biodiversity consequences from land-use change to greater areas of conventional crops because they are being used for bioenergy as well as existing crop uses, but these will depend upon the land-uses that are replaced (Henderson 2009).

Different bioenergy crops present very different habitats for biodiversity and very different changes from previous land-use, notably the contrast between perennial and annual crops, and addition of tall and/or woody structure (Anderson et al. 2004). This means that there are unlikely to be consistent, general patterns of effect across all bioenergy crops.

Notes on TOCB Causality: The Conservation Evidence project has recorded no evidence for bioenergy crop effects (Williams et al. 2020), but several studies have recorded effects for individual crop types, notably short-rotation willow coppice (SRC) and Miscanthus. Miscanthus fields have been found to host a greater abundance and diversity of birds than paired winter wheat fields in both winter and summer (Bellamy et al. 2009). In winter, the greater numbers of birds in Miscanthus fields were probably attracted by the shelter provided by the crop and by the abundance of non-crop plants. During the breeding season, the abundance of non-crop plants in Miscanthus fields, and greater numbers of insects associated with these plants, provided food resources. However, the Miscanthus crop plants provided less insect food than wheat crop plants (Bellamy et al. 2009). The bird effects occurred mostly in the granivorous passerine (excluding Skylark), game bird and snipes groups in winter and were less significant in the breeding season, also being somewhat dependent on the occurrence of non-farmland species such as warblers (Bellamy et al. 2009). Hence, the positive patterns here changes in the species composition of the community as well as some positive effects on species-level abundance. Further, other comparisons of Miscanthus and conventional crops have found no clear differences, probably because Bellamy et al.’s (209) results reflect the weediness and patchiness of the Miscanthus that may not be replicated elsewhere, and the positive effects involve woodland-associated species (in unharvested crops), rather than open farmland species such as Skylark (Sage et al. 2010). Sage et al. also considered SRC, which contained more species and individuals of farmland and woodland birds. However, SRC and Miscanthus will always be an unsuitable breeding habitat for the open-field species that are of conservation concern in farmland and, while these crops can provide habitat heterogeneity that supports increased bird diversity at the landscape scale, this effect would be reduced and would disappear as landscapes became dominated by SRC (Anderson et al. 2004). Moreover, this effect would be greater if bioenergy crops replaced higher-value open-field habitats, such as set-aside fallow (Anderson et al. 2004). Miscanthus may only ever constitute only a suboptimal, supplementary habitat, with only a few birds potentially adapting to use it, and this species set seems to vary regionally (Kaczmarek et al. 2019). Anderson et al. (2004) considered that bioenergy crops would be unlikely to hold significant seed resources to benefit wintering granivores, but, early in its rotation, given suitable management, SRC can feature significant weed densities, providing a valuable seed resource, especially in landscapes lacking other habitats for arable weeds (Fry & Slater 2011).

[TOCB Report-3-3 Soils **ECCM-074**] Targeted introduction of trees, shrubs and scrub to the agricultural landscape is likely to result in an overall reduction in soil erosion risk and a moderate to major positive benefits to soil quality in terms of improved soil structure and increased soil organic matter.

3.4.3.3 Magnitude

Modelling of the potential for carbon sequestration for bioenergy crops in the UK estimated that between -1 Mt CO_2eq and -16 Mt CO_2eq could be sequestered by 2050 under the scenarios considered, in which it was assumed that all *Miscanthus* was grown on cropland, short rotation coppice was split equally between cropland and temporary grassland, and short rotation forestry was split between permanent, temporary and rough grassland in proportion to their coverage in each country in the 2019 June agricultural survey.

It was also assumed that all below ground biomass was lost when crops were replanted, along with 18% loss of any accumulated soil biomass. For more detail about the scenarios considered please see (Thomson et al., 2020).

Current UK bioenergy production is expected to provide 6-18 $MtCO_2$ gross removals per year by 2035 (Element Energy & UKCEH, 2021).

3.4.3.4 Timescale

<5 years for first crop and some fossil fuel abatement
Time until carbon neutrality and net sequestration across the whole life cycle is likely to vary wildly depending on implementation.

3.4.3.5 Spatial Issues

Specifically targeting the area of conversion is necessary to minimise soil loss, N_2O production and displacement of food production.

3.4.3.6 Displacement

There is a strong likelihood that bio-energy crops could displace food production and associated emissions, either domestically or internationally.

3.4.3.7 Maintenance and Longevity

Long-term gains in soil carbon from bio-energy crops would be dependent on continued protection and management that mitigated losses. Emissions that are prevented by avoiding the burning of fossil fuels are reliant on the use of fossil fuels being lower than it otherwise would have been as a result.

3.4.3.8 Climate Adaptation or Mitigation

Any net carbon sequestration will contribute to climate change mitigation.

3.4.3.9 Climate Factors / Constraints

No assessment.

3.4.3.10 Benefits and Trade-offs to Farmer/Land-manager

Income diversification, lower input requirements, potential soil restoration long term.

3.4.3.11 Uptake

Bio- energy crops are being produced in the UK however issues of national capacity have been raised in terms of available area for cultivation due to competition with other land uses (Element Energy & UKCEH, 2021).

3.4.3.12 Other Notes

There is potential to grow bioenergy crops in areas unsuitable for crop production, for example as a result of soil contamination, which would avoid the displacement of food productions whilst potentially providing other ecosystem services at those sites.

3.4.4 ECCM-065: Switch to using peat alternatives in horticultural growing media

3.4.4.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	N
	Below ground carbon sequestration	**L****

The extraction of peat for horticultural use has a significant negative impact on peatlands at the extraction site. Industrial and domestic peat extraction occurring is over approximate 5900 ha and 136600 ha of the UK, respectively (Evans et al., 2017; Brown et al. 2021), although much of the domestic extraction area is inactive (e.g. 88000 ha in Northern Ireland). Peat extraction sites are responsible for the emission +7.9 (domestic use) and +13.8 $tCO_2e\ ha^{-1}yr^{-1}$ (industrial use) (Evans et al., 2017). This is due to the draining of peat and removal of the living layer of vegetation (Gregg et al., 2021). Bare peat is also susceptible to wind erosion but rates of carbon loss from the mechanism have not been quantified (Gregg et al., 2021). Additional off site process and transportation may result in off-site emissions of around 40 t CO_2-C $ha^{-1}\ y^{-1}$ (Gregg et al., 2021). Preventing further peat extraction will preserve the below-ground carbon stores that exist in peatlands.

The exact trade-off of switching to a peat alternative will depend on the alternative that is chosen and a full lifecycle assessment of the replacement material, which are currently lacking. Suggested materials to substitute for peat include coir, green waste compost, decomposed livestock manure, bark, bracken, biochar and hydroponic cultivation.

The majority of research into peat substitutes considers differences in physical and nutritional composition between peat and peat substitutes and their potential to influence plant growth. In gen Li et al. (2009) found that responses to a new medium in root-to-shoot ratio and absolute growth varied depending on the horticultural species being cultivated. Where differences were found, plants grown in cowpeat were associated with a smaller root:shoot ratio than those grown in a peat-based control. They suggest tolerances to the properties of different media could vary substantially across species. A review of another study found no significant differences in yield when growing four vegetable species across 9 different media (Bustamante et al., 2008). However, it is noteworthy that Litterick et al. (2019) report that although performance problems are not widely reported in the scientific literature, consumer platforms and groups in the UK have regularly reported inconsistent or poor performance of peat alternatives for horticultural use. Holmes & Bain (2021) report on 15 successful implementations of peat-free horticulture and discuss the challenges of supplying consistent quality, particularly with new peat-free products.

Despite the evidence gaps around the carbon footprint of peat alternatives, available assessment suggests that switching to alternatives is likely to reduce CO_2eq emissions (Litterick et al., 2019). However, different methods of lifecycle assessment are employed across the range of available products, and certain costs of producing constituent materials are externalised (Litterick et al., 2019). Further comparable research is needed to reach a full conclusion.

3.4.4.2 Co-Benefits and Trade-offs

Greater provision of hydrological services from peatland, and support of biodiversity.
The effect on horticultural productivity at a large scale has not been reliably quantified.

3.4.4.3 Magnitude

If peat extraction is prevented on near natural peat, emissions of +7.9 (domestic use) and +13.8 $tCO_2e\ ha^{-1}yr^{-1}$ (industrial use) could be prevented (Evans et al., 2017).
The lifecycle costs of alternative growing media are not well quantified.
For the effects of restoring sites of current peat extraction, see section 3.3.3.

3.4.4.4 Timescale

< 5 years

Preventing new peat extraction would have an immediate abatement effect.

3.4.4.5 Spatial Issues

Sourcing of alternative media.

3.4.4.6 Displacement

Peatland that is not used for extraction should also not be used for other purposes that result in draining and degradation of peat. The use of peat alternatives that are resource intensive to produce (such as manure), and take material away from bio-energy may have complex consequences to overall carbon emissions and prices if demand in the horticultural industry increases.

3.4.4.7 Maintenance and Longevity

Resuming the use of peat would result in emissions from extraction sites increasing.

3.4.4.8 Climate Adaptation or Mitigation

Any net carbon sequestration will contribute to climate change mitigation.

3.4.4.9 Climate Factors / Constraints

No assessment.

3.4.4.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.4.4.11 Uptake

Peat alternatives are already readily available on the market. However, many are more expensive than peat based growing media and have been associated with less productive growth or variable performance, all of which may prevent uptake. Some substitute such as coir (which is a waste product of the coconut industry) are unlikely to every be produced on a sufficient scale to fully substitute for peat (Litterick et al., 2019). Competition for woodchip and other plant-based substitutes are also raising prices.

The Sustainable Growing Media Task Force (SGMTF) was established by DEFRA in 2010 with an objective of reducing reliance on peat in the sector, which could be leveraged to facilitate commercial uptake (Sustainable Growing Media Task Force, 2012).

3.4.4.12 Other Notes

None

3.5 HABITAT CREATION

Habitat creation is defined as the act of converting one habitat to another, which may or may not have a focus of achieving the properties of a reference habitat. Habitat creation can occur as a part of restoration efforts, where habitat loss has previously occurred. As such, the net carbon balance of any habitat creation is a product of which habitats are converting between (Gregg et al., 2021; Thomas et al., 2020). In addition, the act of converting land use is often associated with a certain rate of carbon emissions, from the process itself and from soil lost due to disturbance. Furthermore, when a non-productive habitat is created on the site of a previously productive one, there is a risk of activity displacement with its own associated emissions, attributable to the original habitat conversion. A recent review by Gregg et al. (2021) summarised the carbon storage potential of 20 habitats and land managements across the UK and is summarised in Figure 1, which is taken from their report.

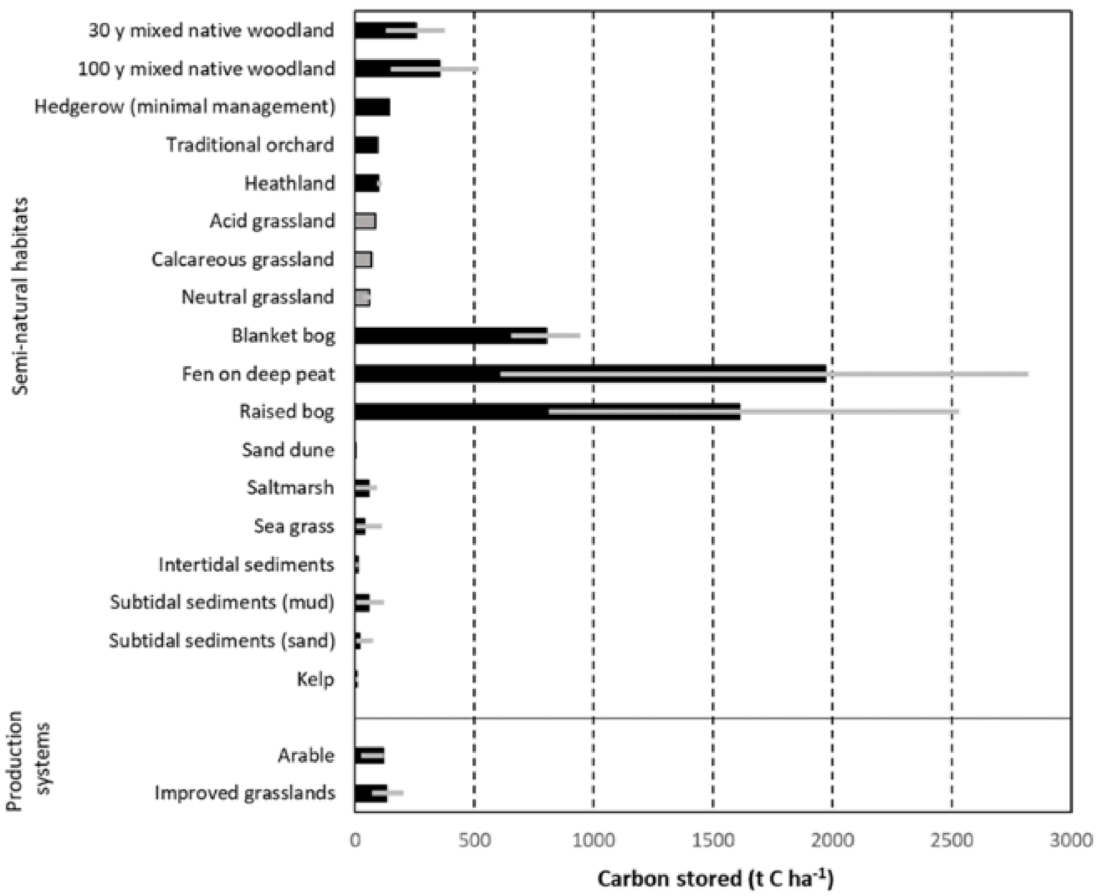

Figure 1: Figure taken from Gregg et al. (2021). The carbon stored across a range of habitats and land managements in the UK using data collected by Gregg et al. (2021). As stated in the original caption, note that semi-natural grassland data are from the top 15cm of soil only and are shown in grey. Other habitats vary the depth for which soil carbon was reported from 15cm to 380cm. Fen data are restricted to deep semi-natural fens. For more information, see the full report.

3.6 HABITAT CREATION - AGROFORESTRY

3.6.1 ECAR-032: Create agroforestry systems

Duplicate evidence base:
Create wood pasture (e.g. through appropriate grazing) [**EBHE-205C**]

3.6.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	T*
Global, regional & local climate regulation	Above ground carbon sequestration	**
	Below ground carbon sequestration	LTD*

Agroforestry is defined as the deliberate integration of woody vegetation within landscapes used for crop and/or animal production, in order to benefit from resulting ecological and economic interactions (Burgess et al., 2018). This definition covers a large diversity of potential practices, including all silvoarable systems (such as intercropping, alley cropping, fallow management, forest farming and boundary planting), silvopastoral systems (such as grazed orchards, woody perennial features combined with livestock and woody perennial features as fodder for livestock), the use of hedgerows and woody boundary features and any combination of the above. In the context of this review we will consider silvoarable systems, silvopastoral systems (Maskell et al., 2019). The creation of hedgerows and woody boundary features will be considered separately in sections 3.9 and 3.13 respectively.

It is well established that woody vegetation can store above and below ground carbon (in both root biomass and soil carbon) long term, with biomass accumulating over the course of many years. The above ground biomass of woody vegetation exceeds the above ground sequestration that can be achieved in grassland or arable systems (Gregg et al., 2021) and is likely to have greater permanence (Matthews, 2020). A study of six agroforestry sites in France (ages 6-41 years) found a mean accumulation rate of 0.65 tC ha^{-1} yr^{-1} for organic carbon in tree biomass (Cardinael et al., 2017). However, changes in soil carbon, particularly compared to grassland, are less well established (Alison et al., 2019; Maskell et al., 2019; Prosser et al., 2022) and in the years after establishment, afforestation is associated with the loss of soil carbon and net GHG emissions. These initial soil losses will ultimately be compensated for given a sufficiently long period, the length of which will depend of growth rate (Eory et al., 2015; Maskell et al., 2019). Trees growing in open locations have reduced competition above and below ground and have been shown to grow faster than densely packed trees. Upson et al. (2016) found that a silvo-pasture system with integrated trees and grassland stored 5% more carbon (above and below ground) than the equivalent area of trees and grassland grown separately, after 14 years. This was in part attributed to the greater size of trees grown in silvo-pasture. Whether difference will persist to greater stand ages is unclear. The optimal distance between trees for maximising carbon sequestration will vary between species and species combinations (Forrester et al., 2017), and may conflict with other management needs for the system. Rapid initial sequestration rates will ultimately slow as stands and trees reach maturity and carbon stocks in vegetation biomass stabilise, subject to management (Matthews, 2020). The length of time required for carbon stocks to reach saturation and maturity are variable between woody species, and sensitive to the local environment, competition and management (Matthews, 2020).

In general, the conversion of land use from less complex to more complex systems is thought to increase soil organic carbon stocks (De Stefano & Jacobson, 2018). (Wiesmeier et al., 2019) suggest that "the storage of SOC increases in the order cropland < forest < grassland", acknowledging some exceptions between forest and grassland. Guo & Gifford (2002) showed that in some cases increases in soil organic carbon can occur when converting from native forest to pasture although they also increase from crop to secondary forest and from pasture to plantations. The conversion of existing forest to agroforestry systems is likely to result in the net emission of carbon, where the clearance of vegetation is required due to the long term reduction in above ground carbon stocks and associated loss of soil carbon (De Stefano & Jacobson, 2018; Matthews, 2020). Estimates of soil organic carbon content between land uses are complex, due to varying distributions of soil carbon with depth and the potential for differing rates of soil loss or soil accumulation across land uses (which are rarely reported in this context) confusing comparisons to depth. Reports of estimated total carbon mass over depth are will be less significantly affected by these discrepancies, but are less commonly reported. Poeplau et al. (2011) found that when grassland was converted to forest, a loss in soil carbon was recorded that increased with the depth of the sample used in analysis. Upson & Burgess (2013) found that increases in soil carbon following the conversion of arable land to a poplar silvo-arable system were significant at 0.6m soil depth but not at 0.2m or 1.5m, 19 years after establishment. A large review of the effects of afforestation on soil carbon by Shi et al. (2013) found that soil organic carbon increase significantly on ex-arable land up to 60cm of depth, but that soil carbon decreased on ex-pastoral land, although not significantly over the full data set. They also found that there was a linear relationship between the rate of change in soil organic carbon in top soil (0-20cm) and deeper soil layers (20-40cm and 40-60cm). Rates of soil carbon change over depths of 0–40 cm, 0–60 cm, and 0–100 cm were 1.33, 1.49, and 1.55 times greater, respectively, than the change at 0–20 cm.

The introduction of woody vegetation to arable systems is thought to have a positive effect of carbon sequestration, in terms of above and below ground carbon stocks overall (Maskell et al., 2019). Prosser et al. (2022) estimate that the conversion of cropland to silvo-arable systems has a high potential for climate change mitigation an estimate a cost effectiveness of £1793 tCO_2^{-1}. Jäger (2017) estimated an additional 0.51 t of SOC ha^{-1} yr^{-1} would be sequestered in the top 25 cm of soil based on 22% of the area being allocated to apple trees, in Swiss silvo-arable systems, where mineral fertilisers and/or slurry where applied to crop row but not to trees. Cardinael et al. (2017) reported mean rates of below ground organic carbon sequestration of -0.24 tC ha -1 yr-1 within the top 30cm of soil -0.65 tC ha -1 yr-1 in the tree biomass in silvo-arable systems. A case study of arable, silvo-arable and forestry systems in the UK estimated via modelling that agroforestry sequestered -127 tCO_2 ha^{-1} over 30 years, whereas arable systems were net emitters of CO_2, accounting for onsite emissions only (Giannitsopoulos et al., 2020). The potential for interactions between woody vegetation and crops has also been discussed, which may reduce the need for inputs, associated with emissions of green-house gases during production and application (fertilisers and pesticides) (Maskell et al., 2019).

Planting trees on pasture for the purposes of silvo-pastoral agriculture is likely to result in the net sequestration of carbon, however the effect on carbon balance due to changes in soil carbon stocks and livestock densities is less certain. Although one study has found carbon stocks in tree biomass to be greater in silvo-pastoral systems than in forest at 99.4 t C ha^{-1}, this was offset by greater losses of carbon stocks in the top 10cm on 6.1 tC ha^{-1} over 14 years (Upson et al., 2016). Prosser et al. (2022) highlight that this loss of soil carbon reduced the overall carbon gain in silvo-pasture by 6%. In a comparison between pasture or silvo-pasture in Northern Ireland, Fornara et al. (2018) found no significant difference in soil carbon and nitrogen stocks in the top 20cm of soil. However, grassland soils contained significantly more macro-aggregates, whilst soils in silvopasture contained significantly more micro-aggregates, which the authors propose may be more stable long term in the face of environmental change. In an experimental silvo-pasture site in North East Scotland soil organic carbon content was similar up to 50cm depth across hybrid larch, scots pine and sycamore, but coniferous species had greater soil carbon stocks in silvo-pasture, whereas sycamore showed the highest soil carbon stocks in woodland. In all cases the pasture control had the lowest soil carbon stocks (Beckert et al., 2016). Intercropping woody vegetation with legumes can reduce cultivation costs and life cycle emissions due to reduced nitrogen inputs. Burgess et al. (2018) report that this practice could support approximately double baseline stocking rates in Spain (0.6 LU ha^{-1} yr^{-1}) without impacting tree growth. However, it should be noted that whilst this may increase profitability, increasing stocking rates is likely to negate a substantial proportion of any carbon gains achieved through agroforestry or reduced fertilised application and could result in net CO_2-eq emissions.

3.6.1.2 Co-Benefits and Trade-offs

No assessment.

3.6.1.3 Magnitude

Cardinael et al. (2017) estimate that SOC stock accumulation in silvo arable systems was 0.24 t C ha^{-1} yr^{-1} at 0–30 cm. C stock accumulation in the tree biomass was 0.65 t C ha^{-1} yr^{-1}.

A meta-analysis found that conversion from pasture/grassland to agroforestry increased SOC stocks by 9-10% (De Stefano and Jacobson, 2018). Upson et al. (2016) found that silvo-pasture systems stored 93.1 t C ha^{-1} over 14 years, including significant losses of soil carbon on establishment.

3.6.1.4 Timescale

>10 years
Timescales are unclear, but the majority of studies reporting net carbon sequestration occur over many years, which in principle allows carbon sequestration in trees to offset losses in soil carbon during land cover conversion.

3.6.1.5 Spatial Issues

Likely to have successful broad implementation, with consideration given to appropriate soil type and species

There is significant variation in the responses of soil carbon to agroforestry that requires further study. Similarly to the growth of woody biofuel crops, targeting degraded soils with lower soil carbon stocks to start with may minimise losses.

3.6.1.6 Displacement

There will be some displacement of agricultural activities if yields cannot be maintained or supplemented by productivity from woody vegetation.

3.6.1.7 Maintenance and Longevity

Woody vegetation may require less maintenance in the form of fertilises and could interact with crops and grassland to reduce the need for pesticides. Carbon stocks in trees have a relatively high residency time (Matthews, 2020). However, preservation of standing carbon stocks active management will be necessary for the persistence of accumulated above and below ground carbon.

3.6.1.8 Climate Adaptation or Mitigation

Accumulated carbon will contribute to climate change mitigation. Agroforestry may also provide a diversification of incomes for landowners and managers.

3.6.1.9 Climate Factors / Constraints

No assessment.

3.6.1.10 Benefits and Trade-offs to Farmer/Land-manager

Adapting to an agroforestry system may be associated with financial and technical barriers to participation, particularly where current methods of mechanised farming will become unsuitable (Prosser et al. 2022). Current rates of agroforestry in the UK are negligible (Thomson et al., 2020).

Potential trade-offs for commercial yields have also been identified, depending on management and valuation of sequestered carbon (Giannitsopoulos et al., 2020), but there is also the potential for additional production from high value trees such as walnut, chestnut, olive and citrus (Burgess et al., 2018).

3.6.1.11 Uptake

No assessment.

3.6.1.12 Other Notes

None

3.7 HABITAT CREATION – COASTAL

The creation of new coastal habitats which are capable of performing the full suite of functions typical of coastal habitats, is a challenge. Coastal ecosystems processes are highly dependent on interactions with the marine environment, and require specific conditions to form. Whilst there is an extensive body of literature concerning the restoration of some coastal habitats that have been lost due to conversion to agriculture, others have almost no evidence base supporting their creation. There are records of artificial coastal habitats being created, such as artificial sand dunes, which are not intended to achieve typical functions of a natural sand dune habitat. Accordingly, most opportunities for coastal habitat creation involve managed coastal realignment, where coastal defences are intentionally breached to allow parts of the terrestrial landscape to become coastal (Macreadie et al., 2019). This may be to increase the national extent of a habitat, or to compensate for habitat losses elsewhere. Facilitate the landwards migration of a coastal habitat with sea level rise, also constitutes habitat creation as any existing coastal land will become marine, whilst previously inland habitats become coastal, although the evidence base for effects of this on carbon sequestration is much smaller. A more general review of the trade-off and potential benefits of managed coastal realignment is provided in section 3.17.1. In this section, we consider the potential for carbon sequestration and storage through the formation of specific habitat types, probable time to formation, and habitat specific challenges to creation.

Coastal and marine ecosystems have been globally recognised as sites of substantial carbon sequestration, through photosynthesis and sediment interception, with the capacity for long term storage in some habitats (Gregg et al., 2021). The services that saltmarsh, sand dunes and machair will provide in the UK between 2020 and 2060 has been estimated to add up to approximate £1 billion by Beaumont et al. (2014). In the context of this report, the coastal habitats we consider for creation are saltmarsh, un-vegetated intertidal systems, sand dunes and shingle, which are within the scope of direct management by ELMs (Burden et al., 2020). However, seas grasses, subtidal sediments and macro-algae are also significant contributors for carbon sequestration and storage, and remain affected by terrestrial land use, despite being beyond the scope of this review (Beaumont et al., 2014; Bertram et al., 2021; Macreadie et al., 2019). Sea grasses in particular are vulnerable to pollution and decrease water clarity, often due to riparian nutrient loading and sediment runoff, which can significantly affect rates of sequestration (Project Seagrass, 2020). A recent review by Parker et al. (2021) provided a best estimate of current blue carbon stocks in the UK, with a summary sediment carbon stocks, standardised to a 1m depth taken from the report shown in Figure 3. This growing understanding of the value of 'blue carbon' comes at a time of increasing pressures and threats to coastal systems from climate change, including temperature rise, increased rainfall, erosion and sea level rise, in addition to pressures directly resulting from human land use such as pollution and grazing pressure. If current rates of coastal habitat loss continue, it is estimated to translate in to the loss of £0.25bn in service value from 2000 to 2060. The creation of new coastal habitats has the potential to offset historic losses to

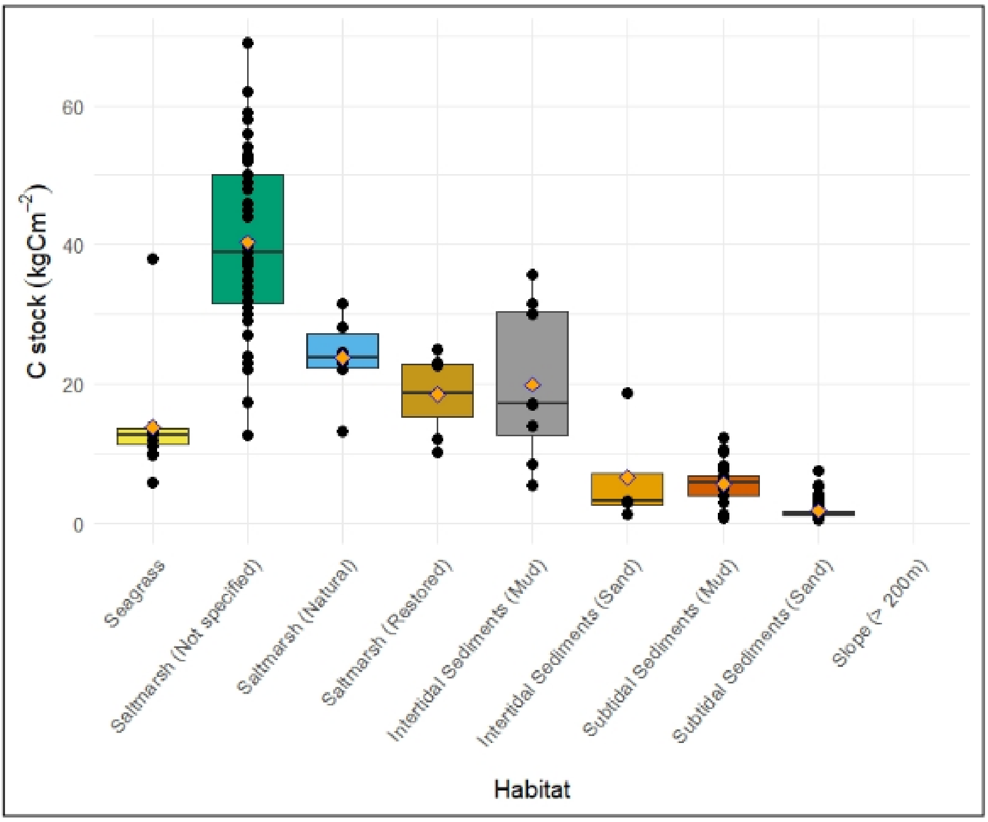

Figure 1 Figure presented in Parker et al. (2021), showing sediment carbon stocks (kg C m^{-2}) for the Secretary of State waters, to a depth of 1m. Minimum, maximum and median quartiles for the data are shown, along with the mean value (orange diamond) for each habitat. Data were derived by a systematic review of the relevant literature. Where carbon stocks were not reported to a depth of 1m, estimates were standardised assuming a uniform distribution of carbon stocks with depth.

alternate land use and mitigate loses to climate change, which may carry benefits for carbon sequestration although trade-offs in other contexts are complex, and direct effects of climate change of rates of sequestration and burial over the next century are unclear (Burden et al., 2020).

3.7.1 EHAZ-070C: Create sand dunes

3.7.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	T*
Global, regional & local climate regulation	Above ground carbon sequestration	LTD*
	Below ground carbon sequestration	LTD*

There is a significant knowledge gap about the potential for sand dunes to migrate inland, in order to preserve this habitat in the face of coastal squeeze. Whilst there may be management actions that could be taken to facilitate this process, such as managing vegetation cover and grazing pressure on the landward edge of coastal dunes (Rural Payments Agency & Natural England, 2020), there is no evidence to indicate whether these measures will be effective in promoting the expansion of dune systems inland. Logic chains suggest that the success of these measures may be highly dependent on physical processes outside of human control, such as wind speed and sediment supply. In particular, there is no evidence for the net effect of these managements on carbon stocks. However, coastal sand dunes are capable of sequestering and storing carbon, as discussed below, where disturbance is low and further research into the potential for promoting sand dune migration is warranted.

Sand dune systems are formed through a delicate balance of physical and biological agents, which has meant that the manual creation of sand dune systems is not something that has been historically possible, although various managements exist to encourage or slow sand dune succession. This sensitivity of sand dunes to factors such as sediment supply, vegetation cover, water table, wave and wind power also make them highly vulnerable to destabilisation and erosion. Dune systems are coupled to surrounding coastal habitats and their sediment budgets have been linked to beach levels and cliff erosion rates (Saye et al., 2006). As such, rising sea levels pose a significant threat to dune systems without sufficient rates of dune migration (Lim et al., 2015; Saye & Pye, 2007). The migration of dune systems (landwards or down drift) (Saye & Pye, 2007) would necessarily involve the conversion of non-dune habitat and continued provision of sufficient sediment and wind force for dune formation to be maintained. The process of dune migration has not been sufficiently studied to comment on the impact of this process on dune carbon stores. However, it is likely that the process with involve some disruption, due to the greater rates of disturbance expected on the existing dune sequence, potentially leading to the initial loss of sequestered carbon. Sea level rise and landwards migration in particular would be expected to affect the height of the water table, which has been associated with significant variation in carbon stock across coastal sand dunes (Jones et al., 2008).

Carbon storage and sequestration rates in coastal dunes have not been well studied in the UK, relative to other coastal systems, however UK specific estimates do exist. Dune habitats are formed by wind-blown particles of sand which accumulate into a succession of ridges that increase in age, stability and vegetation cover with distance from the coastline. Carbon is sequestered by vegetation colonising the dune systems and ultimately sequestered in the sediment. The conditions under which this occurs are highly variable, as a result of variation in moisture content, temperature, nitrogen deposition and grazing pressure, with rates of soil development varying significantly between wetter, low-lying and drier regions (Jones et al., 2008). Beaumont et al. (2014) have estimated the existing area of coastal sand dunes in the England to be 11778 ha (of 71000 in the UK) and to contain a total of 405 tC. Carbon stocks in England's sand dunes were highest in the sediments of dune grasslands (178.7 tC), followed by dune slack (46 tC) and were negligible for more mobile dunes. Vegetation biomass (above and below ground) was also highest in dune grassland (93.7 tC) but was lowest in dune slacks (19 tC). Mobile dunes were estimated to contain 67.5 tC in vegetation biomass. (Jones et al., 2008) report average sequestration rates of -0.582 t C ha^{-1} y^{-1} for dry dunes and -0.73 t C ha^{-1} y^{-1} for dune slacks, resulting in an average rate of -0.595 t C ha^{-1} yr^{-1} (Gregg et al., 2021). This suggests that conversion of grassland habitat may result in a reduction in overall carbon sequestration, even if sand dune establishment is successful.

3.7.1.2 Co-Benefits and Trade-offs

No assessment.

3.7.1.3 Magnitude

The effects of encouraging sand dune formation on carbon stocks are unknown. The carbon stocks of existing coastal sand dunes in the UK have been described by Beaumont et al. (2014) and (Gregg et al., 2021). National carbon stocks for sand dunes in England were highest in the sediments of dune grasslands (178.7 tC), followed by dune slack (46 tC) and were negligible for more mobile dunes. Vegetation biomass (above and below ground) was also highest in dune grassland (93.7 tC) but was lowest in dune slacks (19 tC). Mobile dunes were estimated to contain 67.5 tC in vegetation biomass.

Jones et al. (2008) report average sequestration rates of -0.582 t C ha^{-1} y^{-1} for dry dunes and -0.73 t C ha^{-1} y^{-1} for dune slacks, resulting in an average rate of -0.595 t C ha^{-1} yr^{-1} over a 140 year chrono-sequence of soil development in a temperate dune system in North Wales, where rapid sequestration over the first 60 years was observed, after which accumulation rates dramatically slowed.

3.7.1.4 Timescale

Jones et al. (2008) found that that organic matter accumulation in sand dunes follows a sigmoidal pattern. In dry dunes percentage loss on ignition reached levels comparable to much older dunes 60 years after vegetation establishment. In wet dunes the pattern was found to be similar, but more rapid rates of accumulation were seen, reaching a higher organic matter content.

3.7.1.5 Spatial Issues

Likely to be highly dependent on appropriate physical process to support dune formation and significant questions exist about scalability.

3.7.1.6 Displacement

If successful, formation of new coastal dunes would displace any activity previously existing on that land.

3.7.1.7 Maintenance and Longevity

No assessment.

3.7.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration will contribute to climate change mitigation.

3.7.1.9 Climate Factors / Constraints

Changing weather patterns, water tables and plant productivity are likely affect rates of carbon formation and storage in existing dunes, and any dunes that formed as a result of active management.

3.7.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.7.1.11 Uptake

No assessment.

3.7.1.12 Other Notes

None

3.7.2 ETPW-179C: Create shingle features

3.7.2.2 Co-Benefits and Trade-offs

Global, regional & local climate regulation	Above ground carbon sequestration	**LD***
	Below ground carbon sequestration	**LD***

There are no studies reporting the amount of carbon sequestered by shingle systems in the UK of which we are aware, constituting a significant knowledge gap of coastal system carbon (Beaumont et al., 2014; Parker et al., 2021). Vegetation that develops on shingle will sequester some carbon, but long term storage in this system, is unlikely due to high levels of disturbance and shallow soil layer (Armstrong et al., 2020). Carbon stocks that subsequently enter the marine environment may be sequestered long term (Armstrong et al., 2020).

As a result, the replacement of many habitats (including grassland) with shingle through managed realignment are likely to result in net carbon loss. The replacement of arable land may result in a small amount of net sequestration to the prevention of soil carbon loss (Owens et al., 2006). However, this would be offset by the displacement of agricultural activity.

3.7.3 ETPW-180C: Create inter-tidal and saline habitats

3.7.3.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**T***
	Below ground carbon sequestration	**LT****

Whilst intertidal and saline habitats include vegetated environments, such as shingle and saltmarshes, we consider those habitats separately. In the context of this report, we consider intertidal and saline habitats to refer to non-vegetated systems of mud flats and sand flats that are exposed to the atmosphere at low tide, and which are known to be sites of non-trivial carbon storage in the UK. As such, they were judged to warrant specific consideration.

Intertidal mud and sand flats are primarily anoxic environments that inhibit the breakdown of organic carbon, and as a result are able to sequester carbon deposits long term, potentially for thousands of years (Gregg et al., 2021). The creation of mud flats and sand flats will involve the relaxation or removal of coastal defences and loss of prior land use. Whilst any vegetation cover would be lost in this process, a study of managed realignment in the Humber estuary which created mudflats and saltmarshes suggested an additional 800tC yr^{-1} would be stored as a result over 26 km^2 (0.31 tC ha^{-1} yr^{-1})(Andrews et al., 2008). Carbon that is stored in intertidal sediments may have been initially sequestered in either the marine or terrestrial environment, but in the absence of vegetation these habitats cannot sequester carbon in situ, as such there are complications in quantifying the additionally of carbon sequestered in these habitats. Andrews et al. (2008) also found the created habitats were sequestering nitrogen, phosphorus and heavy metals. Whilst this would remove pollutants from the water column, the long term impact of sediment contamination is unknown.

The current extent of mudflats and sand flats has been estimated at 167 327 ha by Natural England. Intertidal muds (19.9 ± 4.0 SE kg C m^{-2}) tend to contain a higher concentration of carbon that intertidal sands (6.5 ± 0.4 SE kg C m^{-2}), likely due to lower sediment mobility and oxygenation, and a greater surface area of clay minerals for carbon stabilisation (Parker et al., 2021). Parker et al. (2021) estimate that intertidal sediments contain approximately 8.6 million tC across the UK, which is an order of magnitude greater than those in saltmarsh. Data published by Wood et al. (2015) also found that percentage organic carbon varied between

0-7.5% in samples taken from mud and sand flats in Essex and North East England, and that sandier sediments contained less carbon.

Compared to evidence for carbon stocks, the evidence for rates of carbon accumulation is considered poor. Gregg et al. (2021) report average estimates of -1.98 tCO_2 eq ha^{-1} yr^{-1} for the UK, based on a single study for England and a review of Welsh habitats by Armstrong et al. (2020). Parker et al. (2021) identified the same single study of carbon accumulation in muddy sediments at two sites in England by Adams et al. (2012), in their review of the literature, and report mean accumulation of carbon stocks at rates of 83.5 ± 10.2 gC m^{-2} yr^{-1}. Higher rates of carbon sequestration have also been reported in natural mudflats compared to those formed through managed realignment, with average rates of -93.7 gC m^{-1} yr^{-1} and -73 gC m^{-2} yr^{-1}, respectively (Gregg et al., 2021). Rates of sequestration also only account for carbon accumulation, and little is known about net GHG fluxes in these habitats overall (Gregg et al., 2021).

Overall, intertidal sediments can and do sequester large quantities of carbon long term, and as such may be desirable habitats to act as carbon sinks. However, the time frame required for land use conversion to intertidal systems to become a net carbon sink is unknown and will vary with the condition of the land being converted. The source of carbon deposited in intertidal sediments is also unknown and rates of sequestration could vary significantly as a result of environmental context and land use practices. The evidence base for the creation of mud flats and sand flats for carbon sequestration is not yet sufficient to recommend the creation of these habitats on a large scale.

3.7.3.2 Co-Benefits and Trade-offs

No assessment.

3.7.3.3 Magnitude

Sequestration rates of -1.98 t CO_2eq ha $^{-1}$ yr^{-1} and storage potential of 20 tC ha $^{-1}$ in the top 20cm of sediment have been reported, using a combination of observations for England and Wales (Gregg et al., 2021), although this may overestimate what can be achieved through managed realignment.

3.7.3.4 Timescale

>10 years, suggested as the timeframe to be cost effective, although evidence about the time frames for effective restoration arte lacking and require further evidence. (Parker et al., 2021).

3.7.3.5 Spatial Issues

It is unknown how sediment supplies and disturbance rates interact with carbon stores in intertidal sediments.

3.7.3.6 Displacement

May displace prior land use.

3.7.3.7 Maintenance and Longevity

If hard coastal defences are removed, conversion may result in significant financial savings and no long term management would be expected. Carbon sequestered in these habitats is thought to be stored long term, particularly in mud flats. Carbon stocks will be vulnerable to future disturbance.

3.7.3.8 Climate Adaptation or Mitigation

Additional carbon sequestered *in situ* would constitute climate change mitigation. Any long-term storage of carbon transported from elsewhere might also contribute to a reduction in emissions, but this is highly uncertain and not easily quantifiable.

3.7.3.9 Climate Factors / Constraints

No assessment.

3.7.3.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.7.3.11 Uptake

No assessment.

3.7.3.12 Other Notes

None

3.7.4 ETPW-081CX: Create Saltmarsh

3.7.4.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	T*
Global, regional & local climate regulation	Above ground carbon sequestration	*
	Below ground carbon sequestration	**

Assuming creation is attempted at sites with reasonable expectation of saltmarsh formation, such as via managed realignment to restore historic salt marsh or to allow the 'migration' of existing saltmarsh, there is good evidence that facilitating the formation of saltmarsh will lead to carbon sequestration. However, further studies into the potential for methane production and quantification of net GHG fluxes would be beneficial, by providing more supporting evidence.

Saltmarsh has been identified as one of the most important habitats in the UK for blue carbon (Beaumont et al., 2014), and the preservation and expansion of saltmarsh by converting carbon poor terrestrial habitats to saltmarsh is likely to result in the net sequestration of carbon, particularly below ground. Carbon is sequestered *in situ* by salt marsh vegetation and leads to net sequestration and storage in sediments and root structures. In addition, the physical structure of the salt marsh acts as a sink for terrestrial and marine carbon. The sediments of saltmarsh are typically anaerobic, slowing rates of decomposition and resulting in large below ground carbon stocks. Whilst these systems have the potential to store carbon for millennia, disturbance and variable hydrological dynamics mean that periodic remobilisations of sediment can occur (Armstrong et al., 2020). As sea level rise progresses, the risk of saltmarsh loss due to coastal squeeze is significant, without the provision of space to facilitate migration inland (Burden et al., 2020; Oaten et al., 2018) .

Coastal saltmarsh forms in sheltered intertidal systems where vegetation has colonised accumulating sediment but remains subject to tidal flooding (JNCC, 2008). The dynamics of growth (accretion) and erosion vary within a saltmarsh, with variation largely due to the frequency of tidal flooding. Higher sediment turnover and lower carbon contents are typically found at lower elevations and lower turnover and higher carbon content are found at higher elevations where vegetation cover is more developed and diverse (Gregg et al., 2021). Saltmarsh created by managed realignment occurs when existing sea defences are breached to flood reclaimed land. Mudflats quickly form and saltmarsh develops where the elevation is suitable. On shallow sloping sites additional flood defences are required to limit tidal ingress (Element Energy & UKCEH, 2021). The success of saltmarsh creation will be dependent on the existence of appropriate physical and biological conditions, including a positive sediment budget, wave energy and wind exposure, tidal regimes and the availability of appropriate salt tolerant vegetation to colonise the site (Element Energy & UKCEH, 2021). Whilst saltmarsh restoration benefits from the knowledge that required conditions existed in the past, the creation of new saltmarsh from previously inland habitat (particularly as climate change progresses) may be less predictable. Coastal realignment for saltmarsh creation is already underway in the UK, with 2647 ha of saltmarsh created since 1991 (approximately 7-8% of known extent) through managed or unmanaged coastal realignment, although the largest projects are saltmarsh restorations rather than de novo creation (Burden et al., 2020; RSPB, 2021)

Below ground carbon stocks in saltmarsh is some of the highest of any semi-natural habitat in the UK, but values of sequestration rates are highly variable across studies. The current extent of saltmarsh in the UK is estimated between 32100 ha and 38000 ha (Gregg et al., 2021). Total above and below ground (root + soil) carbon stocks in salt marsh have been estimated as 94.7 tC and 4649.6 tC respectively, in England, and 132.0 tC and 5865.8 tC in the UK (Beaumont et al., 2014). In an analysis of existing evidence on blue carbon stocks in Secretary of State waters, Parker et al. (2021) found 82 studies of carbon stocks in UK saltmarsh across natural and restored systems. Natural saltmarshes had an average carbon stock if 23.8 t C ha^{-1} (± 0.2 SE) whilst restored salt marshes had average stocks of 18.6 t C ha^{-1} (± 0.4 SE), and a further 59 studies which did not specify the origin of the saltmarsh had an average stock of 40.3 kg C m^{-2} (± 0.4 SE), all estimated to a standardised depth of 1m. In contrast, Gregg et al. (2021) estimated an average carbon stock of 56 t C ha^{-1} for sediment depths of 10 - 30cm for saltmarsh in England.

As with other blue carbon habitats, data on rates of carbon sequestration or net GHG fluxes are far more scarce than information on carbon stocks. However, there is good agreement that the restoration of saltmarsh previously drained for agricultural use can provide a sustained carbon sink (Burden et al., 2013; Gregg et al., 2021). A global meta-analysis of the effects of coastal management of carbon balance found that carbon stocks in restored saltmarsh generally contained less carbon compared to natural saltmarshes, attributed to the disturbance associated with land use change (O'Connor et al., 2020). Deforested sites restored to saltmarsh showed a particularly high range of soil organic carbon contents compared to natural salt marsh, likely due to variation in forest carbon dynamics, the amount sediment disturbance level and study duration. A study by (Burden et al., 2019) Measured carbon accumulation in sediments with age across nine restored salt marshes in England. Initial rates of carbon stock accumulation were relatively rapid, and indicated a sequestration rate of -1.04 tC ha^{-1} yr^{-1} for the first 20 years, slowing to an average of --0.65 tC ha^{-1} yr^{-1} thereafter. Parker et al. (2021) identified seven studies of carbon accumulation rates in saltmarsh, with one study in natural saltmarsh (-1.185 tC ha^{-1} yr^{-1}), two studies in restored saltmarsh (-0.964 t C ha^{-1} yr^{-1} ± 0.304 SE) and four additional studies in saltmarsh of unspecified origin (-1.606 t C ha^{-1} yr^{-1} ± 0.124 SE). The lowest rate of carbon accumulation found (-0.660 t C ha^{-1} yr^{-1}) was approximately one third of the fasted reported rate (-1.955 t C ha^{-1} yr^{-1}). Gregg et al. (2021) estimated an average rate of carbon burial of -5.19 t CO_2e ha^{-1} yr^{-1} for saltmarsh in the UK. It should be noted that none of these accounts identify carbon stocks or accumulation rates in new saltmarsh, although similarities between the restoration of saltmarsh with a long history of alternate land use might be expected.

Rates of carbon sequestration in saltmarsh are known to be significantly affected by the frequency of submergence, salinity, substrate type, vegetation diversity and pollution (Armstrong et al., 2020). Ford et al. (2016) have suggested that plant diversity positively affects saltmarsh stability due to greater complexity of root structures, and as a result may store carbon for longer than less diverse marshes. This relationship was found to be more important when saltmarsh was composed of less stable substrates such as sand, compared to more stable substrates such as clay. Adams et al. (2012) found that younger saltmarshes at sites of managed realignment had smaller carbon stocks than natural saltmarshes, whilst more developed, realigned sites had carbon stocks that equalled or exceeded those of natural salt marshes. The potential for saltmarsh to sequester carbon long-term is significant due to evidence that carbon stocks do not saturate over decadal timescales, although rates of accumulation slow after 20 years following hydrological restoration of salt marsh (Burden et al., 2019; Element Energy & UKCEH, 2021).

3.7.4.2 Co-Benefits and Trade-offs

Biodiversity, water, pollution, coastal defence, some grazing supported.

3.7.4.3 Magnitude

Modelling by Element Energy & UKCEH (2021) of the potential for greenhouse gas removal (GGR) strategies in the UK suggest a GGR potential of -2.6 to-5.2 tCO_2 ha^{-1} yr^{-1}, with an average of -3.8 t CO2 ha^{-1} yr^{-1} over a 100 time frame for saltmarsh restoration, which may be similar to that of creation. This analysis assumed methane emissions to be zero. They highlight that saltmarsh restoration is a space efficient method, requiring 0.26 ha per tCO_2. At the national scale, they estimate a maximum technical potential for the sequestration of -1.0 Mt CO_2 yr^{-1} by 2050.

3.7.4.4 Timescale

>10 years
Burden et al. (2019) show that rapid carbon accumulation is possible in the 20 years following tidal restoration, but that time frames of close to 100 years might be necessary for equivalency with natural sites to be achieved.

3.7.4.5 Spatial Issues

Targeting is likely critical.

3.7.4.6 Displacement

There is a strong likelihood that land uses (particularly arable agriculture) will be displaced by conversion to saltmarsh. There is some capacity for pastoral agriculture and saltmarsh to co-exist (see 3.7.4).

3.7.4.7 Maintenance and Longevity

Stored carbon is thought to have relatively high longevity in these systems, in the absence of significant disturbance. Given continued sea-level rise, existing salt marsh will become sub-merged, and it is also thought that these carbon stocks are likely to persist relatively long term once beyond the reach of wave action.

3.7.4.8 Climate Adaptation or Mitigation

Any net carbon sequestration will contribute to climate change mitigation.

3.7.4.9 Climate Factors / Constraints

Impacts of sea level rise has been extensively modelled. Latitudinal shifts have not yet characterised
Knowledge of impacts of salinisation on carbonisation are not well known, Thermal effects on carbon cycling arte not well established using empirical data, the effects of climate change on biotic regulation. Increasing wave action may result in some disturbance (Lovelock & Reef, 2020).

3.7.4.10 Benefits and Trade-offs to Farmer/Land-manager

Capital cost for managed realignment are highly variable, with the potential for savings long-term if hard coastal defences are no longer required (Element Energy & UKCEH, 2021).

3.7.4.11 Uptake

The planned release of the UK saltmarsh carbon code may increase willingness to finance or undertake saltmarsh creation projects (UKCEH, 2021b). In general, however, coastal realignment can be met with opposition from the public in some spheres. Managed realignment can be associated with significant up front capital costs, which may deter uptake (Element Energy & UKCEH, 2021).

3.7.4.12 Other Notes

n/a

3.8 HABITAT CREATION – GRASSLAND

3.8.1 ECPW-022C: Create species rich grassland habitats

Duplicated evidence base:

ETPW-205C Create flower-rich and species rich grass margins, field corners, and plots]
EBHE-214C Create locally distinctive flower rich/hay meadows using traditional techniques
EHAZ-010X Create permanent grasslands

3.8.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**T***
	Below ground carbon sequestration	**T***

The impact of creating species rich grassland is variable, and evidence is divided, depending on the prior land use, soil type and subsequent management. In this review we consider the improvement of species diversity to be grassland management, and a discussion of the evidence for the effects of this on carbon stocks and sequestration are provided in 3.11.1. Here, we consider the impacts of converting arable land or agricultural margins to species rich grassland. The conversion of woody habitats or peatlands to species rich grassland is likely to result in the net loss of carbon, which is why the action is associated with a targeted benefit (Evans et al., 2017; Gregg et al., 2021; Matthews et al., in prep.).

There is good evidence to suggest that the conversion of arable land or agricultural margins to diverse grassland can increase rates of carbon sequestration and storage, with increases shown consistently by studies assessing the effects of the incorporation of grasslands into arable rotation (see 3.28.1 and conversion to permanent grassland (Alison et al., 2019; Johnston et al., 2009). Permanent grasslands typically have larger stocks of soil organic carbon compared to arable land as a result of their continuous vegetation cover, additional inputs to the soil via residues and root exudates, and reduced disturbance. Soil carbon data collected in the countryside survey found that the carbon content of arable soils had been declining steadily over 30 years. In the 2007 Countryside Survey, arable soils contained 47 t ha^{-1} in the top 15 cm, whereas improved grassland contained 67.2 t ha^{-1}, neutral grassland contained 68.6 t ha^{-1} and acid grassland contained 90.6 t ha^{-1} (Emmett et al., 2010). In addition, analysis of change in topsoil carbon stocks in the Countryside Survey shows a loss of 11% of soil organic carbon stocks (tC ha^{-1}) from 1978 to 2007 in arable soils (Emmett et al. 2010), whilst diverse grassland systems have the potential to be net sinks for GHGs (Gregg et al. 2021). DEFRA (2007) found that small changes in land use to grassland, limited to arable field margins, resulted in a significant increase in soil carbon stocks. The incorporation of legumes and nitrogen fixing species has been found to be particularly effective at increasing rates of carbon sequestration in grassland (Alison et al., 2019; Gregg et al., 2021). However, it should be noted that the initial conversion to grassland, any cutting, fertilising and additional reseeding will be associated with carbon emissions (Element Energy & UKCEH, 2021). If permanent grassland is grazed, them emissions attributed to the livestock should also be incorporated into assessments. Evidence about the long-term magnitude of improvements to carbons stocks that can be achieved through conversion to a diverse grassland, and changes in soil carbon at depth are also lacking.

A significant caveat to this that the conversion of arable land on peat to grassland is likely to have significantly smaller benefits in terms of net GHG emissions abatement than peatland restoration (see 3.3).

The conversion of wetland habitats, scrubland and woodland to species rich grassland carry significant risks for rates of carbon sequestration and storage. In wetland habitats, associated drainage and disturbance can result in significant carbon emissions from below ground carbon stores (see 3.3). In habitats with more significant above ground carbon, conversion will involve the loss of these carbon stocks and future sequestration potential (see other sections in 3.5 for various potentials for sequestration and storage that may exceed that of diverse grassland). There is uncertainty about the effect of grassland cover on soil carbons stocks compared to woodland cover, and comparisons are sensitive to land use history, management and surveyed soil depth. However, the overall potential for carbon sequestration of woodland is likely to be significantly higher than that of grassland (Gregg et al., 2021; Matthews, 2020).

3.8.1.2 Co-Benefits and Trade-offs

No assessment.

3.8.1.3 Magnitude

Falloon et al. (2004) suggests that 0.1-2.4% of the UK's 1990 CO_2 emissions could be sequestered via scaling up the planting diverse grassland on arable margins.

Larger scale conversion of arable land to grassland has been estimated to result in an increase in SOC of 20 tC ha^{-1} over approximately 50 years (Johnston et al., 2009), and the accumulation of additional soil carbon at a rate of 1.01 t C ha^{-1} y^{-1} (Conant et al., 2001).

3.8.1.4 Timescale

< 5 years
As reported by Alison et al. (2019) rate of SOC accumulation on newly formed grasslands increased with plant diversity over 4 years in a study by Steinbeiss et al. (2008).

3.8.1.5 Spatial Issues

Sward mixes are likely to need tailoring to local conditions to be successful. Countryside Stewardship schemes have tested for outcomes on sward diversity, and targets are often not met.

3.8.1.6 Displacement

Arable production will be displaced, particularly if productive regions are converted to grassland.

3.8.1.7 Maintenance and Longevity

Reseeding may be required relatively frequently maintain desired species and diversity levels, depending on the management of the grassland and grazing intensity, which is likely to be associated with the loss of SOC to some extent.

It is unclear how long carbon stocks can be expected to increase for, following conversion of arable land to grassland, but stocks are expected to ultimately saturate (Johnston et al., 2009).

3.8.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration will contribute to climate change mitigation.

3.8.1.9 Climate Factors / Constraints

Grassland systems have a low to medium sensitivity to climate change according to Gregg et al. (2021), with sensitivity varying across types of grassland and management. Extended periods of drought can affect productivity and in extreme cases may cause grasses to die back. Some research has suggested more diverse communities are more resilient to drought event, in small scale manipulations (Isbell et al., 2015).

3.8.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.8.1.11 Uptake

No assessment.

3.8.1.12 Other Notes

None

3.9 HABITAT CREATION – HEDGEROW

3.9.1 ECCM-025C: Plant hedgerows

Duplicate evidence base: Plant hedgerows around point-source polluters [**ECCM-080C**]; **EBHE-191** Plant and establish appropriate species of field boundary trees

3.9.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	*
Global, regional & local climate regulation	Above ground carbon sequestration	**
	Below ground carbon sequestration	L*

Hedgerows are linear woody features in agricultural landscapes of human creation that are typically composed of tree and shrub species, and are usually actively managed to limit outwards growth (Gregg et al., 2021), and may be associated with additional linear features such as walls, fences or ditches. The 2007 Countryside Survey reports 477,000 km of hedgerows in Great Britain and 402,000 km in England and showed a 6.2% reduction in the length of managed hedgerows in Great Britain between 1998 and 2007 (Carey et al., 2008).

The potential for the expansion of hedgerows to contribute to the UK's net zero goals is widely acknowledged, but relatively few studies have been conducted on rates of sequestration and storage in hedgerows, rather than using estimates from agricultural or woodland settings (Gregg et al., 2021). A 30-40% increase in the extent of hedgerows estimated to lead to between a change in emissions of -1 and -4 Mt CO_2eq by 2050, assuming some of the extent is managed for biofuel production, when considered in concert with the effects of agroforestry (Thomson et al., 2020). Above ground carbon stocks reported in Gregg et al. (2021) vary between 26.7 tC ha $^{-1}$ and 131.5 tC ha^{-1}, across a variety of species compositions and managements in the UK. Axe et al. (2017) conducted one of the largest studies to date on hedgerow carbon stocks in the UK, and showed the significance of management on hedgerow carbon stocks. Hedgerows that were maintained in taller and wider states contained more carbon in above ground biomass, with a 4.2m wide hedge containing 9.7 tC km^{-1} more than a hedge 2.6m wide, with a mean height of 3.5m. When biomass is removed during management, sequestering that carbon in biochar or using biomass to reduce the use of fossil fuels through biofuel production offer methods of preserving net GHG reductions attained through carbon sequestered in hedgerows (Gregg et al., 2021).

In addition to carbon sequestered above ground, hedgerows have the potential to increase carbon stocks below ground, through root biomass, exudate production, increased rates of litter deposition, and reduced rates of soil disturbance from mechanised farm operations (Gregg et al., 2021). Axe et al. (2017) reported that almost 50% of carbon stocks may be below ground, and that roots and below ground woody debris accounted for an additional 38.2 tC ha^{-1}. As discussed in Gregg et al. (2021), further research has suggested that hedgerows established for biodiversity, with higher species richness, have higher below ground carbon stocks from 0-1m (175.9 ± 13.2 t C ha^{-1}) than remnant hedgerows (132.7 ± 7.3 t C ha^{-1}). Hedgerows can also reduce rates of soil erosion when established as contour features and have been associated with a progressive increase in soil depth on their downwards slopes (Gregg et al., 2021). Initially establishment of hedgerows may be associated with a loss of soil carbon in the short term, based on the agroforestry literature (see 3.6).

Notably less is known about rates of carbon sequestration in hedgerows and how quickly they can be expected to decline with hedgerow age. Understanding of woodland dynamics suggests that less regular pruning will result in higher carbon stocks but slower rates of carbon sequestration (Matthews, 2020). Rates of sequestration with be highest in young hedgerows, with Robertson et al. (2012) estimating that new hedgerows sequestered soil organic carbon at a rate of -0.54 t C ha^{-1} yr^{-1} whereas old hedgerows sequestered -0.46 t C ha^{-1} yr^{-1}, and took hundreds of years to reach an equilibrium SOC stock, based on modelling using data taken from woodland understory dynamics.

Overall, establishing hedgerows along field margins within agricultural landscapes is highly likely to result in net carbon sequestration, and increase carbon stocks above and below ground. However, there are significant evidence gaps about the effects of hedgerow composition and rates of carbon sequestration under various management regimes (Gregg et al., 2021).

3.9.1.2 Co-Benefits and Trade-offs

Biodiversity
Soil protection

[TOCB Report-3-5D Systems **EBHE-191**] Assuming that 'appropriate' precludes planting of trees in open landscapes where they might cause negative effects on open-country species, this can only have positive effects on biodiversity, especially relative to 'inappropriate' trees, such as non-native species.

3.9.1.3 Magnitude

Above ground carbon stocks in hedgerows reported in Gregg et al. (2021) vary between 26.7 t C ha^{-1} and 131.5 t C ha^{-1}, across a variety of species compositions and managements in the UK, but are known to increase with hedgerow width and height.

Below ground carbon stocks may account for half of the overall carbon stock of hedgerows, with Axe et al. (2017) reporting that roots and below ground woody debris accounted for below ground stocks 38.2 tC ha^{-1} Rates of sequestration above and below ground are largely unknown, and most current estimates are from woodland systems.

A 30-40% increase in the extent of hedgerows estimated to lead to between a change in emissions of -1 and -4 Mt CO_2eq by 2050, assuming some management for biofuel production, when considered in concert with the effects of agroforestry (Thomson et al., 2020).

3.9.1.4 Timescale

> 10 years.
Rates of sequestration are not well established in the literature. Below ground carbon stocks may take multiple hundreds of years to saturate, based on one modelling study (Robertson et al., 2012). Literature from agroforestry systems suggests net carbon stocks may take multiple years to become positive.

3.9.1.5 Spatial Issues

Likely to have successful broad implementation, with consideration given to appropriate soil type and species

Little is known about the interactions between hedgerow establishment and soil organic carbon in the short term, and how that might vary with baseline carbon stocks.

3.9.1.6 Displacement

No assessment.

3.9.1.7 Maintenance and Longevity

Hedgerows require regular maintenance to maintain their structure, and after approximately 40 years without structural maintenance, will lose their structural density, likely resulting in a loss of carbon stocks. Shorter term management and trimming is usually carried out on a scale of 1-3 years (Gregg et al., 2021).

3.9.1.8 Climate Adaptation or Mitigation

Carbon sequestered by hedgerows constitutes climate change mitigation.

3.9.1.9 Climate Factors / Constraints

No assessment.

3.9.1.10 Benefits and Trade-offs to Farmer/Land-manager

Hedgerows have ongoing management needs, which carries a financial burden. However, hedgerows may confer indirect benefits to land managers as a result of ecosystem services supported.

3.9.1.11 Uptake

No assessment.

3.9.1.12 Other Notes

Hedgerow creation has the potential to support national capacity for biochar and biofuel production (Thomson et al., 2020)

3.10 Habitat Creation – Mountain, moor and heathland

3.10.1 ECPW-176C: Create heathland (including heathland mosaics)

3.10.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**LT ***
	Below ground carbon sequestration	**LT ***

This review considers the creation of heathland, which is a structurally diverse habitat of conservation value, associated with varying intensities of management including grazing and burning to maintain desired community compositions and habitat functions. As discussed in 3.1.2, the creation of heathland on peat soils is associated with significant carbon emissions and is strongly advised against for the purpose of carbon sequestration. In most cases, heathland creation is likely to occur on grassland, which may have replaced heathland in the past as a result of intensive grazing or burning regimes, (Gregg et al., 2021). The felling of woodland or scrub to create heathland would also result in the loss of above and below ground carbon stocks, and is not considered further in this review (Matthews et al., in prep.).

There is good evidence that overall rates of carbon sequestration are high in heath and moorland communities, to the extent that result at rates of sequestration (-3.45 t C ha^{-1} yr^{-1}) are comparable to the lower end of what is seen in of UK woodland (Gregg et al., 2021; Quin et al., 2014). Most evidence comes from comparing carbon stocks and sequestration rates in either restored or target heathland with those of grassland on historic or degraded heathland sites, with some evidence from encroachment on alpine grassland (Gregg et al., 2021).

There is reasonable agreement across the literature that the majority of heathland carbon stocks occur in soils, with as little as 2% of carbon stock in vegetation biomass (ONS, 2020; Gregg et al., 2021). The 2007 Countryside Survey estimated that England's dwarf shrub heath contained approximately 81.6 t C ha^{-1} in sediments (0-15cm) (Carey et al., 2008). A review of carbon stocks in UK heathland found reported mean below ground carbon stocks to vary between 88 tC ha^{-1} and 103 t C ha^{-1} across 4 studies, and mean vegetation carbon stocks of 2 tC ha^{-1}, 7.11 tC ha^{-1}, 9 tC ha^{-1} and 49 tC ha^{-1} across 4 studies (Gregg et al., 2021). The lowest (50.7 tC ha^{-1}) and highest (196 tC ha^{-1}) individual measurements for soil carbon stocks reported by (Cantarello et al., 2011). Carbon stocks in heathland soils and rates of GHG emissions are highly dependent on management regimes (burning, drainage of wetland sites and grazing patterns) and this variation should be taken into account when considering the creation of heathland and its intended future management (Gregg et al., 2021).

3.10.1.2 Co-Benefits and Trade-offs

Biodiversity. No assessment.

3.10.1.3 Magnitude

Quin 2014 (as reported by Gregg et al., 2021) showed that upland heath established on acid grassland contained 100.16 ± 5.66 tC ha^{-1} in soil and 12.1 ± 0.8 tC ha^{-1} in vegetation, compared to 102.01 tC ha^{-1} (±4.10) and 12.0 tC ha^{-1} (± 0.6) in target upland heath, with soil carbon assessed to 15cm depth. Rates of sequestration in upland heath were estimated to be -3.45 t C ha^{-1} yr^{-1}.

3.10.1.4 Timescale

Time to achieve soil carbon stocks typical of target heathland will likely depend on the history of the site, condition of soil prior to creation and management applied to heathland.

3.10.1.5 Spatial Issues

Avoidance of heathland creation on peatland is critical.

3.10.1.6 Displacement

Some displacement of grazing may occur to maintain appropriate intensity for the heathland ecosystem.

3.10.1.7 Maintenance and Longevity

Heathland requires ongoing management to persist.

3.10.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration will contribute to climate change mitigation.

3.10.1.9 Climate Factors / Constraints

Prolonged summer drought condition in wet upland heath have been shown to convert wet mineral soils under heathland from net carbon sinks to net carbon sources (Reinsch et al., 2017; Sowerby et al., 2010).

3.10.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.10.1.11 Uptake

No assessment.

3.10.1.12 Other Notes

None

3.11 Habitat Creation – Scrub

3.11.1 EBHE-203C: Create targeted scrub

3.11.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	T*
Global, regional & local climate regulation	Above ground carbon sequestration	L**
	Below ground carbon sequestration	LT**

Although there is a robust logic chain suggesting that the creation of scrubland in targeted areas should result in below and above ground carbon sequestration, measurements of scrub carbon stocks and sequestrations rates for the UK are lacking in the literature.

In the UK, the creation of scrub typically occurs as part of secondary succession from agricultural land taken out production, although it can occur as part of primary succession on bare ground associated with quarries, dunes and scree. Scrub typically occurs as part of the transition from herbaceous and grassland vegetation to woodland, but definitions that distinguish scrub from the two alternate habitats are variable and largely arbitrary (Mortimet et al., 2000). Here we use the definition suggested by Mortimet et al. (2000) which includes all stages of developing scrubland from scattered bushed to closed canopy vegetation, typically less than 5m tall, which may include native shrubs, non-native shrubs and saplings, with a few scattered mature trees. This definition serves to include carr, upland and lowland scrub, woodland edge habitats, montane and coastal scrub, but excludes coppice regrowth, young planted trees and dwarf shrub heath. As such, the term scrub can refer to a large variety of species, some of which have high conservation value (such as juniper) and some which are considered undesirable or invasive (such as rhododendron).

In some cases, scrubland can rapidly establish with little human assistance, and creation may simply require a reduction in management intensity. In other cases, particularly where appropriate seed sources are absent, creation may be a more active process including planting, protection of developing scrub, managing grazing pressure and weeding (English Nature & RSPB, 2003). Some level of grazing or cutting is required to maintain scrub and prevent further succession and encroachment on other habitats of conservation value. However, the creation of scrub may have additional carbon sequestration value as part of assisted natural regeneration towards woodland (see 3.16).

There has been little research into the carbon content of scrub within the UK, although the accumulation of carbon through agricultural land abandonment and subsequent scrub development has been extensively studied globally, with a number of studies from Europe. Gregg et al (2021) reflect that “there have been studies on the direction of secondary succession after abandonment or neglect of priority habitats but not many which have focused on the carbon implications of either allowing scrub to colonise other habitats or the management to contain it. There seems to be a specific evidence gap in the UK”. In a global review of the effect of different land management options following agricultural abandonment, Bell et al. (2020) found that a combination for active and passive management for restoration or rewilding gave the largest increases in soil carbon, although responses were highly geographically variable. A review of European land abandonment found that positive ecosystem service outcomes included increased soil carbon and reduced soil erosion, although increased risks of fire and soil erosion on hilly areas were identified as potential negative consequences (Ustaoglu & Collier, 2018).

Differences between communities and environmental conditions represented in the literature and those anticipated in the UK mean that reported rates of carbon storage and sequestration are unlikely to be representative. As woody vegetation, it is logically consistent that scrubland carbon dynamics will be similar to those of young woodland, accumulating above and below ground carbon stocks, although initial planting (if required) may result in a short term loss of soil carbon stocks (sections on 'Habitat Creation – Woodland' and on 'Woody Features'). The establishment of scrub has been shown to increase soil organic content due to high rates of litter deposition and nitrogen enrichment, where nitrogen fixing species are part of the community (Mortimet et al., 2000), however further study of UK scrubland carbon dynamics are desirable.

3.11.1.2 Co-Benefits and Trade-offs

No assessment.

3.11.1.3 Magnitude

Unknown, but expected to be between values expected for diverse grassland and woodland reported elsewhere in this report.

3.11.1.4 Timescale

Unknown, but stabilisation of soil carbon stocks is likely to take >10 years, following evidence from woodland, hedgerow and agroforestry systems.

3.11.1.5 Spatial Issues

Targeting is key.

3.11.1.6 Displacement

The creation of scrub on agricultural land will displace some agricultural activity, particularly if the area is highly productive and intensively managed. There is capacity for scrub to be grazed in a silvo-pastoral context, which may minimise risks of displacement, but over grazing will likely negatively impact carbon sequestration potentials.

3.11.1.7 Maintenance and Longevity

Scrub will require long term management to maintain in the majority of climatic conditions in the UK, and prevent expansion into habitats of conservation or agricultural importance. However, preventing succession to woodland may not be desirable in the context of maximising carbon sequestration potential.

3.11.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration will contribute to climate change mitigation.

3.11.1.9 Climate Factors / Constraints

No assessment.

3.11.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.11.1.11 Uptake

Grants may already be available to landowners to facilitate the planting of scrub as part of woodland creation. An opinion survey of land managers in England and Wales found that 87% or respondents thought some scrub was a nuisance, but in the majority of cases, the area that was considered a nuisance was less than 10% of the known area (Mortimet et al., 2000).

3.11.1.12 Other Notes

n/a

3.12 Habitat Creation – Woodland

3.12.1 EBHE-104: Create a woodland creation plan

3.12.1.1 Causality

Global, regional & local climate regulation	Above ground carbon sequestration	**T**
	Below ground carbon sequestration	**T**

For a more detailed assessment of the potential of woodland creation see section on 'Habitat Creation – Woodland'.

Although the effects of using a woodland creation plan on carbon sequestration have not been evaluated in the literature, there is a consensus opinion that detailed consideration of the objectives behind tree planting, appropriate methodology, location, and the development of an adaptive management plan are strongly associated with successful tree planting and initiatives (Brancalion & Holl, 2020). The consideration of these issues is also park of the UKFS. The value of increasing woodland carbon stocks for climate change mitigation is contingent on the preservation of those stocks as sequestered carbon long term (Matthews, 2020). Strategies, such as using a woodland creation plan, increase the likelihood of long term woodland persistence are therefore likely to be beneficial for carbon sequestration. However, the complex trade-offs involved in woodland establishment, between land uses, water resource regulation, carbon sequestration biodiversity and cultural values are complex, and specialist expertise is likely needed to appropriately evaluate plans in a context dependent manner. The consideration of long-term woodland management strategy and putting in place an adaptive management plan are desirable components of a woodland management plan (Brancalion & Holl, 2020), as management has a significant effect on the resilience of woodland to future pressures and it's carbon sequestration and storage potential (Matthews, 2020). However, in the absence of financial support for long term management, planning may not yield rewards.

It is a legal requirement in the UK for an Environmental Impact Assessment to be conducted and approved by the Forestry Commission when a proposed afforestation project exceeds a minimum threshold in scale (0.5ha), or when adjacent to recent afforestation (Environmental Impact Assessments for woodland - GOV.UK (www.gov.uk)). In addition, woodland Creation Planning Grants in England are contingent on the development and submission of a woodland creation design template, and completion of this plan can be supported via an application for a £1000 grant (Woodland Creation Planning Grant - GOV.UK (www.gov.uk)). Forests planned are required to be compliant with national legislation on forest standards (UKFS). Whilst it is beyond the scope of this review to comment on existing legislation, it is of note that neither process requests information on planned management or long-term security of woodland, which are processes of significant importance for carbon sequestration and storage capacity. In an assessment of barriers to tree planting in England, when asked what actions would be most beneficial to overcoming regulatory barriers to tree planting, 45% of respondents said reducing the time and cost associated with the Environmental Impact Assessment for afforestation would be beneficial (DEFRA, 2021b). The top three financial barriers identified across approximately 1400 responses were:

1. Widening of the criteria of woodland creation grants

2. Higher payments for incentives for woodland creation and
3. Introducing mechanisms to release a secure long term cash flow for ecosystem services.

This suggests a delicate balance exists between achieving high rates of tree planting and woodland creation, whilst ensuring the creation of high quality woodland that has a positive effect on ecosystem services and is viable long term.

3.12.1.2 Co-Benefits and Trade-offs

Biodiversity.

3.12.1.3 Magnitude

The impact of using a woodland creation plan on carbon sequestration and storage has not been quantified. Matthews et al. (in prep.) presents and analysis of carbon sequestration and storage over time as a result of afforestation in nine study sites with varied compositions and managements. The estimated rate of carbon sequestration in living woody biomass during a 100 year period from the time of woodland creation was in the range -1 to 3 -tC ha^{-1} y^{-1} (-3.7 to -11 tCO_2 ha^{-1} y^{-1}), and commonly takes a value around -1.4 tC ha^{-1} y^{-1} (-5 tCO_2 ha^{-1} y^{-1}). This summary excluded stands managed for clear felling.

Matthews et al. (in prep.) also report model estimates for 11 contrasting options for woodland creation involving tree species mixtures and management practices relevant for the UK. These options include natural recolonisation of non-wooded land with broadleaf trees and 'light' subsequent management, examples of mainly moderate- and fast-growing commercial coniferous woodlands, and managed woodlands composed of complex tree species mixtures. The mitigation contributed by the woodland options contrasts in magnitude over time, with different options providing the most mitigation benefits at different times (between 2022 and 2100 and beyond) and in different ways (involving direct carbon sequestration or GHG emissions savings through provision of wood products, to varying extents). It should be noted that the different woodland options are not always interchangeable on the same site or in the same location within the UK. Planting 1 hectare of woodland in 2022 is estimated to result in net carbon sequestration in woodlands and wood products over the period from 2022 to 2050 of between 0.8 and 13.8 tCO_2 ha^1 y^1, with a mean estimate for all 11 options of 5.7 tCO_2 ha^1 y^1. If GHG emissions savings arising from utilisation of additional wood products and wood fuel are allowed for, these estimates increase to between 1.7 to 32.0 tCO_2 ha^1 y^1, with a mean estimate for the 11 options of 12.4 tCO_2 ha^1 y^1. Matthews et al. also report estimates of mitigation per hectare for a programme of woodland creation based on the 11 example woodland options over 25 year period starting in 2022. Such a programme is estimated to provide net carbon sequestration of between 0.16 and 4.9 tCO_2 ha^1 y^1, with a mean estimate for all 11 woodland options of 1.6 tCO_2 ha^1 y^1. (Hence, for example, a programme to create 20,000 ha of woodland over 25 years would provide carbon sequestration of between 3 and 98 $ktCO_2$ y^1, with a mean estimate of 32 $ktCO_2$ y^1 for a combination of the woodland options). If GHG emissions savings arising from utilisation of additional wood products and wood fuel are allowed for, the per-hectare estimates for the 25 year programme are 0.12 to 10.7 tCO_2 ha^1 y^1, with a mean estimate for all 11 options of 3.3 tCO_2 ha^1 y^1.

3.12.1.4 Timescale

The impact of a successful woodland creation plan would be felt at a range of timescales, by (for example) increasing the likelihood of successful forest establishment, minimising negative consequences from land use change, and supporting the long term condition of the woodland.

3.12.1.5 Spatial Issues

No assessment.

3.12.1.6 Displacement

No assessment.

3.12.1.7 Maintenance and Longevity

The long term support of management and monitoring for created woodland is key to the maintenance of carbon stocks.

3.12.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration will contribute to climate change mitigation.

3.12.1.9 Climate Factors / Constraints

No assessment.

3.12.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.12.1.11 Uptake

Dedicating sufficient time and resources to an effective woodland creation plan carries a cost that is likely to require compensation, acknowledged by the current government scheme.
The availability of financial support to enable long term management plans to be put in place and followed is also important to translate a plan into a successful outcome.

3.12.1.12 Other Notes

None

3.12.2 EBHE-281: Set up or engage with community tree planting projects

3.12.2.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**LT *** **
	Below ground carbon sequestration	**LTD ****

There is strong evidence supporting the positive effect of planting trees on carbon sequestration above and below ground, but we found no evidence for the effect of community driven tree planting on carbon sequestration and storage, compared to other methods of establishment. A study of urban tree survival rates in the US found no significant effect of who planted the tree on mortality rates, out of contractors, organisation staff and volunteers (Wattenhofer & Johnson, 2021).

The effect of tree planting on carbon sequestration can vary enormously with species, location and management practice. Any community tree planting project should be appropriately supported by specialist advice, to ensure appropriate species and site selection, and the provisioning of appropriate management after the fact. Trade-offs with other ecosystem services of value can also occur as a result of tree planting that should be considered. For a more detailed review of the impacts of woodland and tree planting see other actions in sections on ‘Habitat Creation – Woodland’ and on ‘Woody Features’.

Community tree planting projects can be a cost effective method of increasing forest cover, and can promote community engagement with nature and a sense of wellbeing (Turner-Skoff & Cavender, 2019). However, it is well known that suitable long term management has a significant role in the carbon storage potential and sequestration rate of planted trees, including survival rates (Matthews, 2020; Widney et al., 2016)

(Wattenhofer & Johnson, 2021). Therefore, community tree planting projects should be supported by provisions for long term adaptive management and access to appropriate expertise and resources to be successful.

3.12.2.2 Co-Benefits and Trade-offs

Public wellbeing and engagement with nature can be enhanced by planting trees in area with public access and through public involvement (Turner-Skoff & Cavender, 2019).

3.12.2.3 Magnitude

Matthews et al. (in prep.) presents and analysis of carbon sequestration and storage over time as a result of afforestation in nine study sites with varied compositions and managements. The estimated rate of carbon sequestration in living woody biomass during a 100 year period from the time of woodland creation was in the range -1 to -3 tC ha^{-1} y^{-1} (-3.7 to -11 tCO_2 ha^{-1} y^{-1}), and commonly takes a value around -1.4 tC ha^{-1} y^{-1} (-5 tCO_2 ha^{-1} y^{-1}). This summary excluded stands managed for clear felling.

Matthews et al. (in prep.) also report model estimates for 11 contrasting options for woodland creation involving tree species mixtures and management practices relevant for the UK. These options include natural recolonisation of non-wooded land with broadleaf trees and 'light' subsequent management, examples of mainly moderate- and fast-growing commercial coniferous woodlands, and managed woodlands composed of complex tree species mixtures. The mitigation contributed by the woodland options contrasts in magnitude over time, with different options providing the most mitigation benefits at different times (between 2022 and 2100 and beyond) and in different ways (involving direct carbon sequestration or GHG emissions savings through provision of wood products, to varying extents). It should be noted that the different woodland options are not always interchangeable on the same site or in the same location within the UK. Planting 1 hectare of woodland in 2022 is estimated to result in net carbon sequestration in woodlands and wood products over the period from 2022 to 2050 of between -0.8 and -13.8 tCO_2 ha^{-1} y^{-1}, with a mean estimate for all 11 options of -5.7 tCO_2 ha^{-1} y^{-1}.

If GHG emissions savings arising from utilisation of additional wood products and wood fuel are allowed for, these estimates increase to between -1.7 to -32.0 tCO_2 ha^{-1} y^{-1}, with a mean estimate for the 11 options of -12.4 tCO_2 ha^{-1} y^{-1}. Matthews et al. also report estimates of mitigation per hectare for a programme of woodland creation based on the 11 example woodland options over 25 year period starting in 2022. Such a programme is estimated to provide net carbon sequestration of between -0.16 and -4.9 tCO_2 ha^{-1} y^{-1}, with a mean estimate for all 11 woodland options of -1.6 tCO_2 ha^{-1} y^{-1}. (Hence, for example, a programme to create 20,000 ha of woodland over 25 years would provide carbon sequestration of between -3 and -98 $ktCO_2$ y^{-1}, with a mean estimate of -32 $ktCO_2$ y^{-1} for a combination of the woodland options). If GHG emissions savings arising from utilisation of additional wood products and wood fuel are allowed for, the per-hectare estimates for the 25 year programme are -0.12 to -10.7 tCO_2 ha^{-1} y^{-1}, with a mean estimate for all 11 options of -3.3 tCO_2 ha^{-1} y^{-1}.

3.12.2.4 Timescale

See section on 'Habitat Creation – Woodland'.

3.12.2.5 Spatial Issues

No assessment.

3.12.2.6 Displacement

No assessment.

3.12.2.7 Maintenance and Longevity

The long-term support of management and monitoring for created woodland is key to the maintenance of carbon stocks.

3.12.2.8 Climate Adaptation or Mitigation

There is strong evidence that planting trees can result in net GHG sequestration, constituting climate change mitigation. Planting trees that are genetically diverse and selected for resilience to pests, warmer conditions and climatic extremes will promote the adaptation of English forests to climate change.

3.12.2.9 Climate Factors / Constraints

No assessment.

3.12.2.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.12.2.11 Uptake

No assessment.

3.12.2.12 Other Notes

None

3.12.3 ECCM-048: Create woodland on a large scale

Duplicated evidence:
Plant large-scale woodland in priority catchments [**ECCA-018C**]; Create targeted woodland [**ECPW-044C**]; Use woodland management (UKFS) for target priority woodland species [**ETPW-266**]

3.12.3.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T*****
Global, regional & local climate regulation	Above ground carbon sequestration	*****
	Below ground carbon sequestration	**TD ****

There is very strong evidence that the creation of woodland can result in significant carbon sequestration above and below ground, following best practices. Woodlands have the most rapid potential rates of carbon sequestration of any habitat and are associated with substantial carbon stores both above and below ground (Gregg et al., 2021; Matthews et al., in prep.; Poulton et al., 2003; Prosser et al., 2022).

Rates of carbon sequestration in woodland are initially rapid and will slow over time, due to reduced rates of primary productivity at the stand level and increased respiration rates, reflecting the saturation of the carbon sink (Coomes et al., 2012; Gregg et al., 2021; Matthews, 2020). The carbon stocks and sequestration rates of contemporary woodland can serve as a guide for what might be achieved through woodland creation, and have been well studied, with the carbon sequestration potentials of many species available in the Woodland Carbon Code (Home - UK Woodland Carbon Code). In a review of carbon dynamics across semi-natural and managed habitats in the UK, Gregg et al. (2021) estimate average soil and vegetation carbon stocks that could be achieved for mixed, native broadleaved woodland at 30 and 100 years of age (see Table 1).

Estimates of contemporary woodland carbon stocks the UK were made by Matthews et al., in prep. and are reproduced in Table 2, but exclude any stocks in harvested wood products. Matthews (in prep) found that across eight long term afforestation study sites, all sites saw a significant increase in total carbon stocks, compared to those under the pre-existing land use. These forests included those established by natural colonisation and planting, a range of thinning treatments and were composed of oak, mixed broadleaves, Scots pine or Sitka spruce (with a 45 year clearfelling rotation).

Table 1 Reported values of soil and vegetation carbon stocks and sequestration rates in mixed, native broadleaved woodland in the UK (un-thinned), reproduced from data presented by Gregg et al. (2021).

Woodland Age (years)	Soil carbon in 0-15 cm (t C ha^{-1})	Soil carbon in 0-100cm (t C ha^{-1})	Vegetation carbon (t C ha^{-1})	Carbon flux, averaged over age (t CO_2eq ha^{-1} yr^{-1})
30	55 (50 to 59)	151 (108 to 173)	114 (22 to 204)	-14.5 (-2.5 to -25.5)
100	55 (50 to 59)	151 (108 to 173)	203 [41 to 344)	-7 (-2 to -13)

Table 2 Estimates of mean carbon stocks per hectare in woodlands in the different countries, taken from Matthews et al. in prep, for which estimates were derived using statistics in Forest Research (2020) and woodland areas provided by Brown et al. (2021).

	Woodland carbon stocks			
Carbon pool	England	Wales	Scotland	Northern Ireland
	Carbon stock (tC ha^{-1})			
Above-ground	62	57	58	51
Below-ground	22	21	21	19
Total for living trees	85	78	79	70
Deadwood	11	15	13	19
Litter	15	19	16	27
Soil	159	343	197	522
Total woodland carbon stocks	270	455	305	639
	Carbon stock (tCO_2 ha^{-1})			
Above-ground	228	211	213	186
Below-ground	82	76	77	71
Total for living trees	311	286	290	257
Deadwood	41	55	47	71
Litter	54	69	60	100
Soil	584	1 256	723	1 914
Total woodland carbon stocks	990	1 667	1 120	2 343

Rates of change in soil carbon stocks following afforestation are highly variable in the literature, with contradictory patterns observed with depth and over time (Gregg et al., 2021). Prosser et al. (2022) estimate that planting forests can initially result in the loss of approximately 2.0 tCO_2 ha^{-1} yr^{-1} as a result of site preparation, and that losses can remain significant over a 30 year time horizon. However, variation in processes of establishment, underlying variability in soil type and even tree species planted cause significant variation on rates of carbon loss from soil (Bárcena et al., 2014; Laganière et al., 2010). Losses can be particularly high on soils rich in organic matter where soils are ploughed or drained as part of site preparation (Gregg et al., 2021, Matthews et al., in prep.).

Matthews (in prep.) found that the highest rates of early net carbon loss occurred when planting trees with slow growth rates on organo-mineral soils, and when site preparation techniques involved significant soil disturbance. In one experimental site (a higher yield Sitka Spruce plantation) Matthews et al. (in prep.) report annual net gain in soil carbon stocks by around year 20, at which time the total net loss of soil carbon since initial ground preparation (but not including losses due to road construction) had reached 19.7 tC ha^{-1}. In a notably shorter study (four years) in North Wales, Ahmed et al. (2016) found no significant effect on tree planting or species composition on soil carbon content across four depths between 0 and 100 cm, with the exception that less carbon was present in the top 10cm of *Fagus sylvatica* plots, although this could be due to a plot level effect or initial disturbance during planting. In most cases on arable soils, rapid sequestration by young stands would be expected to achieve net sequestration after a small number of years, due to the accumulation of carbon in vegetation biomass, and an increase in soil carbon in arable soils (Bárcena et al., 2014; Gregg et al., 2021; Laganière et al., 2010). Practices that minimise soil erosion, and avoidance of sites with high baseline levels of carbon as advised in the UKFS will help minimise this risk.

Achieving the highest possible rates of carbon sequestration in woodland is dependent on selecting the best species for the target environment, an approach which is often referred to as "right tree, right place, right reason". Recent evidence reviews by Matthews et al., in prep. and (Gregg et al., 2021) conclude that there is no single best woodland structure for creation in the UK, and different woodland compositions and managements are likely to support different suites of functions.

Coniferous species can produce more harvestable timber due to higher rates of biomass allocation to their trunk, and (under optimal conditions) are capable of faster growth rates than broadleaf species in the first 30 years (when contributions will have the most impact towards the achievement of net zero by 2050). However, native broadleaved species sequester more carbon per unit volume of wood, and can sequester carbon at similar rates to conifers over longer time periods (Gregg et al., 2021). Matthews et al., in prep. reports that over a 100 year time span, broadleaf woodland managed for conservation / environmental objectives can sequester similar carbon to commercial managed coniferous woodlands of typical growth rate, at the per hectare level. However, the highest rates of sequestration per hectare can be achieved by commercially managed coniferous woodland with yield classes significantly above the UK mean, assuming appropriate environmental conditions.

After harvesting, carbon stocks present in the woodland significantly decrease although harvested wood products may continue to contribute towards carbon sequestration. In most cases, the choice of woodland composition will be determined the wider functions the woodland is intended to fulfil and the limitations of the environment in which establishment can occur (Matthews et al., in prep.). For example, the best forest composition for rapid carbon sequestration is unlikely to be the same as the best composition for securing large carbon stocks, long term (Figure 4, taken from Matthews et al 2021). There is evidence that a diverse forest canopy and understory can maximise carbon stocks per unit area, due to a more efficient use of space by tree crowns, but evidence for interactions between large numbers of species is rare (Matthews, 2020). Diverse forests are also likely to have a better outcome for biodiversity, and increase forest resilience to pests and disease long term (see discussion in section 3.4).

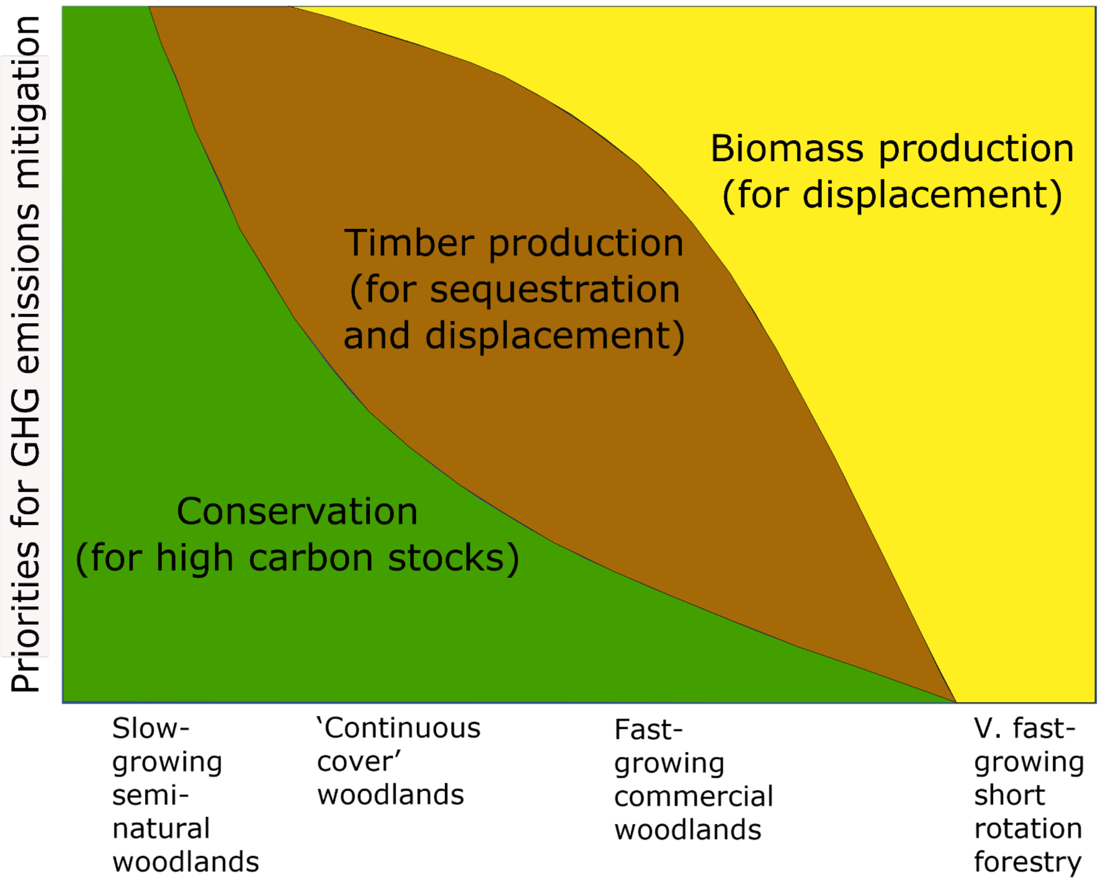

Figure 2 Taken from Mattews 2021 illustrating the anticipated relationship between woodland creation options and GHG mitigation potential, as a result of potential growth rates of trees (which is a product of both species selection and site). Some site properties may be best suited to one type of woodland over another.

Consideration of long term management strategy is vital when evaluating the carbon sequestration potential of woodland creation. When managed for conservation purposes, carbon stocks in woodland could be secured for centuries in the absence of large disturbances (Figure 4, taken from Matthews et al 2021). However, managed forests are associated with the intermittent removal of carbon stocks to various degrees through thinning or harvesting. In addition to soil carbon loss through disturbance, removed carbon stocks then enter an alternate trajectory for decomposition which varies according to intended use (Matthews et al., in prep.). For example, burning harvested biomass results in the immediate return of most carbon to the atmosphere, whereas incorporation into furniture and construction can extend the residency of sequestered carbon by decades (Matthews, 2020; Matthews et al., in prep.). Due to this potential lag, the accumulation of carbon stocks can occur outside of the forest system.

Matthews et al. (in prep. and 2020) also demonstrate how (assuming continuous management) the growth of new woodland for clear felling is associated with a non-trivial long term average carbon stock, that is particularly apparent when averaging over multiple managed woodlands at the landscape scale, although mean living biomass stocks will be lower than those for old growth broadleaf woodland (Figure 3) (Gregg et al., 2021; Matthews, 2020). Trade-offs in wood production are complex, and harvested wood products may also substitute for the use of fossil fuel derived materials, providing additional emissions abatement when considering the full product lifecycle (Matthews, 2020; Matthews et al., in prep.). Lastly, it is also important to note that product life cycles and emissions associated with site preparation, establishment, management, harvesting and transport can be significant, to the extent that forestry can become a net source of carbon in extreme cases with low yield species (Matthews, 2020), and so considering the future management of newly created forests from the outset is critical to achieving a desired outcome (Matthews, 2020; Matthews et al., in prep.).

The sequestration and storage potential of woodland effectively scales with area (assuming appropriate conditions for growth and establishment). Therefore, the potential for storage and sequestration per hectare is not materially different for targeted or large-scale woodland creation, although management options and needs may vary. There is evidence that trees at the edge of woodlands show slightly higher growth rates relative to the interior (McDonald & Urban, 2004). However, in the long-term, larger woodlands are likely to have greater stability and resilience to disturbances, including wind damage and temperature variation (Besliu et al., 2022, Hallinger et al. 2016). Woodland size and location are also known to significantly impact the relationship between woodland, biodiversity, soil health and hydrology (see co-benefits).

3.12.3.2 Co-Benefits and Trade-offs

Not assessed.

3.12.3.3 Magnitude

The maximum potential carbon uptake from woodland creation in the UK is site dependent. The largest carbon storage possible over 50 years is estimated to be by Sitka Spruce in the North and South-West of England, approaching 1450 tC ha^{-1}. However, aspen and sycamore are expected to provide higher rates of sequestration in much of the midlands and East of the country (reported by Gregg et al., 2021, using data from the Forestry Commission and analysed by Forest Research).

Using data from the woodland carbon code, Gregg et al. (2021) estimate sequestration rates for un-thinned, mixed, native broadleaved woodland as shown in Table 1. Over a 30 year period, they also report representative values for sequestration by beech (-11.5 t CO_2eq ha^{-1} yr^{-1}) and oak (-15.5 t CO_2eq ha^{-1} yr^{-1}) but state that woodlands of beech or oak are likely to be less resilient to climate change.
Matthews et al. (in prep.) presents and analysis of carbon sequestration and storage over time as a result of afforestation in eight study sites with varied compositions and managements. The estimated rate of carbon sequestration in living woody biomass during a 100 year period from the time of woodland creation was in the range -1 to -3 tC ha^{-1} yr^{-1} (-3.7 to -11 tCO_2 ha^{-1} yr^{-1}), and commonly takes a value around -1.4 tC ha^{-1} yr^{-1} (-5 tCO_2 ha^{-1} yr^{-1}). This summary excluded stands managed for clear felling.

Matthews et al. (in prep.) also report model estimates for 11 contrasting options for woodland creation involving tree species mixtures and management practices relevant for the UK. These options include natural recolonisation of non-wooded land with broadleaf trees and 'light' subsequent management, examples of mainly moderate- and fast-growing commercial coniferous woodlands, and managed woodlands composed of complex tree species mixtures. The mitigation contributed by the woodland options contrasts in magnitude over time, with different options providing the most mitigation benefits at different times (between 2022 and 2100 and beyond) and in different ways (involving direct carbon sequestration or GHG emissions savings through provision of wood products, to varying extents).

It should be noted that the different woodland options are not always interchangeable on the same site or in the same location within the UK. Planting 1 hectare of woodland in 2022 is estimated to result in net carbon sequestration in woodlands and wood products over the period from 2022 to 2050 of between -0.8 and -13.8 tCO_2 ha^{-1} yr^{-1}, with a mean estimate for all 11 options of -5.7 tCO_2 ha^{-1} yr^{-1}. If GHG emissions savings arising from utilisation of additional wood products and wood fuel are allowed for, these estimates increase to between -1.7 to -32.0 tCO_2 ha^{-1} yr^{-1}, with a mean estimate for the 11 options of -12.4 tCO_2 ha^{-1} yr^{-1}. For more information, see the full report.

Matthews et al. (in prep.) also report estimates of mitigation per hectare for a programme of woodland creation based on 11 contrasting example woodland options over 25 year period starting in 2022. Such a programme is estimated to provide net carbon sequestration of between -0.16 and -4.9 tCO_2 ha^{-1} yr^{-1}, with a mean estimate for all 11 woodland options of -1.6 tCO_2 ha^{-1} yr^{-1}. (Hence, for example, a programme to create 20,000 ha of woodland over 25 years would provide carbon sequestration of between -3 and -98 $ktCO_2$ yr^{-1}, with a mean estimate of -32 $ktCO_2$ y^{-1} for a combination of the woodland options). If GHG emissions savings arising from utilisation of additional wood products and wood fuel are allowed for, the per-hectare estimates for the 25 year programme are -0.12 to -10.7 tCO_2 ha^{-1} yr^{-1}, with a mean estimate for all 11 options of -3.3 tCO_2 ha^{-1} yr^{-1}.

Thomson et al. (2020) estimate that up between 279 kha (business as usual) and 2050 kha (widespread engagement) could be used for the planting of new forest by 2050 across all land use scenarios. For more information on assumptions and requirements for the various scenarios considered, see the full report.

Estimates of the total potential for atmospheric greenhouse gas removal from the afforestation of both conifer and broadleaf woodland and forest management in the UK were -3 to -5 Mt CO_2 yr^{-1} in 2030, and -16 to -24 Mt CO_2 yr^{-1} in 2050 by a review of greenhouse gas removal options in the UK (Element Energy & UKCEH, 2021). These figures assume the afforestation of 0.3-0.6 Mha by 2030, 0.8-1.3 Mha by 2040, and 1.4-2.0 Mha by 2050, following projected land use change and residual land availability under the Committee on Climate Change (CCC) 6th Carbon Budget (CB) Land-based scenarios (Thomson et al., 2020). These figures do not include areas planted in agroforestry or forest management outside of that specified in the CCC 6th CB scenarios. The report estimates an average CO_2 uptake of -13.2 tCO_2 $ha^{-1}yr^{-1}$ by 2050, with a maximum technical potential for atmospheric greenhouse gas removal of -26.5 $MtCO_2$ in 2050, under the CCC 6th CB Balanced Net Zero pathway, and assuming 100% utilisation of available residual land.

3.12.3.4 Timescale

>10 years
Woodlands tend to only become net carbon sinks some time during the first few decades of growth, and may take up to 30 years to store significant amounts of carbon. Therefore, only woodlands planted in the near future (or with very high growth rates) will have a material impact on the achievement of net zero by 2050 (Gregg et al., 2021; Matthews et al., in prep.).

Matthews et al. (in prep). also present estimates of changes in carbon stocks and sequestration rates across eight case study forests for 100 years after establishment under varied management regimes and compositions. After 45 years most sites show a carbon stock increase between approximately 50 and 150 tC ha^{-1} across all woodland carbon pools (living biomass, litter, deadwood, soil). The site with the smallest increase in stocks was associated with large soil carbon loss. Fast growing coniferous plantations can accumulate carbon stocks of over 300 tC ha^{-1} up to the time of harvesting (assumed here to involve clearfelling), after which carbon stocks are diminished by harvesting and longer-term contributions to carbon sequestration are very limited. A longer-term contribution to mitigation by commercially managed woodlands depends on how harvested wood is utilised, i.e. to ensure carbon is retained in wood products and that wood products effectively displace GHG-intensive materials and fuels.

3.12.3.5 Spatial issues

Woodland creation can be carried out over large areas of the England, but the appropriate targeting of woodland composition will have a significant impact on the magnitude of carbon sequestration possible, and may be critical for tree survival.

3.12.3.6 Displacement

Woodland creation, particularly over a large scale, will cause displacement of prior land uses either internationally or nationally.

3.12.3.7 Maintenance and Longevity

The longevity of carbon in woodland is largely determined by management practices. In the absence of management, carbon stocks can be relatively long term (hundreds of years), but when faced with disturbance carbon can be lost from woodland systems relatively quickly.

The preservation of carbon stocks that are developed in subsequent years by woodland creation will be largely contingent on future generations guaranteeing those stocks.

3.12.3.8 Climate Adaptation or Mitigation

Carbon sequestered in woodland constitutes climate change mitigation.
Methods of woodland creation could also be used which would result in woodlands that are more resilient to climate change.

Woodlands can also provide refugia from climate change for biodiversity through microclimate buffering, where conditions within a forest can be several degrees cooler than outside (De Frenne et al., 2019).
Forests have a lower albedo than many other land cover types, and as a result they reflect less energy from the earth' surface. There is some concern that planting forests over large area at a global scale will reduce land surface albedo and reduce the efficacy of CO_2 sequestration (Ellison et al., 2017).

3.12.3.9 Climate Factors / Constraints

Woodland carbon stores and sequestration rates are potentially vulnerable to climate change. Extreme weather conditions have the potential to result in high rates of CO_2 loss from woodlands and supress rates of sequestration, and can cause large areas to become temporary carbon sources (Ciais et al., 2005). More generally, the ability of woodlands to maintain rates of productivity and survival under climate change and increase risk of international disease and pest transmission are a cause for concern, given the potential reversibility of carbon sequestration by woodland.

3.12.3.10 Benefits and Trade-offs to Farmer/Land-manager

Additional benefits may be accrued due to harvesting material from woodland or the sale of carbon credits, supported by the woodland carbon code. Woodlands also carry amenity value and can promote a sense of wellbeing. Planting woodland may remove significant areas form traditional agricultural practices, with associated consequences for income.

3.12.3.11 Uptake

A survey of barriers that prevent woodland expansion conducted by (DEFRA, 2021b)found that the top three financial barriers identified across approximately 1400 responses were:
1. Widening of the criteria of woodland creation grants,
2. Higher payments for incentives for woodland creation and
3. Introducing mechanisms to release a secure long term cash flow for ecosystem services.

Key non-financial barriers identified in the report revolved around access to specialist knowledge or guidance and resources, such as diverse or locally adapted seed sources.

3.12.3.12 Other Notes

None

3.12.4 ECPW-071C: Create floodplain woodland

3.12.4.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	T**

Global, regional & local climate regulation	Above ground carbon sequestration	L***
	Below ground carbon sequestration	LTD**

There is strong evidence that woodland creation can result in the net sequestration and storage of considerable carbon over the next century, at a national scale (see section on 'Habitat Creation – Woodland'). However, there is little evidence specific to woodlands on floodplains in the UK. Floodplain woodland is defined as all woodland within the fluvial floodplain that is subject to intermittent flooding (Environment Agency, 2017). Although woodlands on floodplains could store significant carbon, their promotion above and beyond woodland in other locations is primarily driven the range of other ecosystem services that floodplain woodlands are expected to deliver, including but not limited to flood protection, soil stabilisation, and benefits for biodiversity (although evidence for some of these benefits is mixed) (Brown et al., 1997; Iacob et al., 2014; Lawson et al., 2018; Nisbet & Thomas, 2008). The density woodland and area used for planting are likely to vary depending on the relative priority given to the various functions floodplain forests could perform, with large-scale unbroken forests likely to be undesirable for biodiversity (Brown et al., 1997).

There is strong evidence from afforestation practices more generally that replacing arable land or grassland with forest will result in above ground carbon sequestration and storage (see section on 'Habitat Creation – Woodland'). Average carbon stocks in above ground biomass, deadwood and litter for hardwood stands in the active floodplain of the Elbe, Germany, have been estimated as 50.2 tC ha^{-1} (± 10.8 SE) in young plantation, 140.6 tC ha^{-1} (± 11.6 SE) in old, dense stands and 180.4 tC ha^{-1} (± 26.6 SE) in old sparse stands (Shupe et al., 2021). A global review of carbon sequestration in riparian forests found a median biomass carbon stock value of 63 tC ha $^{-1}$, over 491 observation at 53 study sites, and a wide range of forest ages (Dybala et al., 2019).

Planted forests contained significantly more carbon in biomass for their age than naturally established riparian forests. Riparian woodlands were also associated with significantly higher carbon stocks than control sites, (Dybala et al., 2019). A case study in California found that restoration of woody vegetation in a riparian floodplain resulted in an increase in soil carbon stocks at a rate of 0.87 tC ha^{-1}yr^{-1}, whilst vegetation carbon stocks were modelled as 279.3 tC ha^{-1} after 50 years (Matzek et al., 2020) Increases in carbon stocks are likely due to a combination of in situ carbon sequestration and storage, increased retention of particulate organic matter from flood waters, and reduced rates of topsoil erosion. Without knowledge of what percentage of increase soil carbon stocks constitutes in situ sequestration, it is challenging to assess the additionality of such an increase in carbon stocks, although increased litter inputs and living biomass accumulation make net sequestration likely.

In the absence of afforestation, hydrologically restored floodplains can store significant quantities of carbon (Cook, 2007; Gregg et al., 2021; Sutfin et al., 2016). This is likely to be partially offset by methane emissions, which have be documented in wetlands due to microbial respiration in anoxic conditions, (IPCC Task Force on National Greenhouse Gas Inventories, 2014). Finally, we know of no studies quantifying changes in the rates of GHG fluxes from floodplain soils with afforestation, although this area of uncertainty is considered unlikely to offset potential benefits from sequestration that are previously discussed..

3.12.4.2 Co-Benefits and Trade-offs

No assessment.

3.12.4.3 Magnitude

Evidence for the carbon storage potential of UK floodplain woodlands is generally lacking, although the carbon storage potential of woodland more generally is well known – see 3.12.3.

Average carbon stocks in above ground biomass, deadwood and litter for hardwood stands in the active floodplain of the Elbe, Germany, have been estimated as 50.2 tC ha^{-1} (± 10.8 SE) in young plantation, 140.6 tC ha^{-1} (± 11.6 SE) in old, dense stands and 180.4 tC ha^{-1} (± 26.6 SE) in old sparse stands (Shupe et al., 2021).

3.12.4.4 Timescale

>10 years
No reported rates of carbon sequestration in flood plain forests could be found for this review, or how those rates change with age. Evidence from other forest systems suggests that forests become net carbon sinks in the 10+ years following establishment, due to carbon debts from soil disturbance, but evidence from floodplain systems is also lacking.

3.12.4.5 Spatial Issues

No assessment.

3.12.4.6 Displacement

Afforestation in the floodplain will displace agricultural activities that may be taking place there, unless displacement can be avoided as part of an agroforestry system.

3.12.4.7 Maintenance and Longevity

The longevity of carbon in woodland is largely determined by management practices. In the absence of management, carbon stocks can be relatively long term (hundreds of years), but when faced with disturbance carbon can be lost from woodland systems relatively quickly.

The preservation of carbon stocks that are developed in subsequent years by woodland creation will be largely contingent on future generations guaranteeing those stocks.

Additional carbon stored in floodplain soils could represent a stable long-term store, if the stability of floodplain sediment is maintained.

3.12.4.8 Climate Adaptation or Mitigation

Carbon sequestered in woodland constitutes climate change mitigation.
Methods of woodland creation could also be used which would result in woodlands that are more resilient to climate change.

3.12.4.9 Climate Factors / Constraints

Tree species that are likely to thrive in hypoxic, wet floodplain soils are likely to be more vulnerable to cavitation and drought stress, and sensitivity may be greater in young stands (Shupe et al., 2021). Increasing climatic variability may put forests adapted to floodplain conditions at greater risk.

Woodland carbon stores and sequestration rates are potentially vulnerable to climate change. Extreme weather conditions have the potential to result in high rates of CO_2 loss from woodlands and supress rates of sequestration, and can cause large areas to become temporary carbon sources (Ciais et al., 2005). More generally, the ability of woodlands to maintain rates of productivity and survival under climate change and increase risk of international disease and pest transmission are a cause for concern, given the potential reversibility of carbon sequestration by woodland.

3.12.4.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.12.4.11 Uptake

No assessment.

3.12.4.12 Other Notes

None

3.12.5 EBHE-209C: Create traditional orchards with local varieties of fruit tree

Duplicate: **ECCM-055** Plant traditional orchards

3.12.5.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**L****
	Below ground carbon sequestration	**LD***

Reports of carbon storage a sequestration in English orchards are scarce, however net carbon sequestration may occur, assuming orchards are not replacing mature trees or scrub. A review of carbon storage and sequestration by habitat in the UK estimated that traditional orchards can support soil carbon contents of an average of 73.75 t C ha $^{-1}$ to a depth of 30cm (with a range of observed values between 47 and 111 t C ha^{-1}). Vegetation in traditional orchards is estimated to contain an average of 21.4 t C ha^{-1} (with reported range of 8.6 to 230.4 t C ha^{-1}) (Gregg et al., 2021). Both mean values are associated with a low degree of confidence due to a lack of evidence in the literature (one published study was identified; Robertson et al., 2012) and variable planting patterns and tree densities. The same study reported average rates of carbon sequestration of -2.89 t CO_2eq ha^{-1} yr^{-1} (range: -5.89 to +1.65 t CO_2eq ha^{-1} yr^{-1}) in traditionally managed orchards. Interestingly, although traditional orchards were reported to maintain higher carbon stocks, intensively managed orchards sustained higher rates of carbon sequestration due to higher rates of pruning and stand density.

Variation in the rate of sequestration (or in some cases emission) from traditional orchards is likely due to reduced rates in primary productivity with age, combined with high rates of carbon removal each season in fruit biomass and the loss of material through pruning. Material that is removed from orchard is reportedly often burnt, returning much of the sequestered carbon to the atmosphere. As a result of these outgoing fluxes, orchards have a net sequestration rate (not including the emissions associated with the processing of produce or management practices) smaller than other types of woodland and can result in net carbon emissions (Gregg et al., 2021; Matthews, 2020).

3.12.5.2 Co-Benefits and Trade-offs

Biodiversity.

3.12.5.3 Magnitude

A review of carbon storage a sequestration by habitat in the UK estimated that traditional orchards can support soil carbon contents of an average of 73.75 t C ha $^{-1}$ to a depth of 30cm (with a range of observed values between 47 and 111 t C ha^{-1}). Vegetation in traditional orchards is estimated to contain an average of 21.4 t C ha^{-1} (with reported range of 8.6 to 230.4 t C ha^{-1}) (Gregg et al., 2021).

3.12.5.4 Timescale

>10 years
Traditional orchards are suggested to reach a carbon equilibrium between 50 and 100 years of age (Gregg et al., 2021).

3.12.5.5 Spatial Issues

No assessment.

3.12.5.6 Displacement

May displace agricultural production depending on placement and whether land use change occurs. If converting intensive orchards to traditional orchards, production may be displaced elsewhere naturally due to a decrease in yields.

3.12.5.7 Maintenance and Longevity

The longevity of traditional woodland suggests a moderately secure carbon stock, in the presence on continued management without intensification that typically results in the loss of carbon stocks.

3.12.5.8 Climate Adaptation or Mitigation

Any net carbon sequestration will contribute to climate change mitigation.

3.12.5.9 Climate Factors / Constraints

Woodland carbon stores and sequestration rates are potentially vulnerable to climate change. Extreme weather conditions have the potential to result in high rates of CO_2 loss from woodlands and supress rates of sequestration, and can cause large areas to become temporary carbon sources (Ciais et al., 2005). More generally, the ability of woodlands to maintain rates of productivity and survival under climate change and increase risk of international disease and pest transmission are a cause for concern, given the potential reversibility of carbon sequestration by woodland.

3.12.5.10 Benefits and Trade-offs to Farmer/Land-manager

Landowners may commercially benefit from fruit trees, unless converting from a more productive form of agriculture.

3.12.5.11 Uptake

No assessment.

3.12.5.12 Other Notes

Modifications to traditional practice, such using dead material and pruned biomass for biochar of biofuel production could favourably affect the carbon balance of traditional orchards.

3.13 HABITAT CREATION – WOODY FEATURES

3.13.1 ECCM-024: Plant or manage trees outside of woodlands, including shelterbelts

Duplicated evidence base:
Plant trees alongside water courses to provide shade and reduce water temperatures within rivers [**ECCA-036**]; and Create wind breaks [**ECPW-080C**] and Plant trees and shrubs around point-source polluters [**ECPW-156C**]; Plant/ manage trees to slow water, particularly cross-slope planting [**ECCA-017**]; **EBHE-303**: Plant trees and hedges to mitigate the visual impact of polytunnels from the immediate view of neighbouring residential dwellings; **EBHE-273**: Plant/ manage trees and shrubs to mitigate noise from transport and facilitate positive sound;

3.13.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**
	Below ground carbon sequestration	**TD***

The evidence base for the benefits of the creation of woody features outside of woodlands for climate regulation is largely composed of the evidence for the creation of hedgerows and agroforestry systems, which is described in detail elsewhere in this review.

There is strong evidence that planting trees can result in the net sequestration of significant above and below ground carbon at decadal time scales, under appropriate soil and climatic conditions, for the target tree species (Gregg et al., 2021; Matthews, 2020). Young trees achieve the highest rates of carbon sequestration, with growth rates slowing with age as rates of respiration increase. Although rates of sequestration are lower in larger trees, they represent valuable carbon stocks, particularly ancient or veteran trees (Gregg et al., 2021). Veteran trees can have substantial cultural and biodiversity value, and establishing the next generation of veteran trees is necessary to preserve this resource long term (Read, 2000). More generally, linear woody features and isolated trees could be a valuable store of genetic diversity and contribute towards the overall resilience of UK tree populations to pressures of climate change, disease and pests. The current area of tree cover outside of NFI woodland in England (defined as areas with <0.5ha extent and <20m width) is thought to be 565.0 kha or 4.3% of land area (compared to 1336.4 kha within NFI woodland) (National Forest Inventory, 2017). This area of tree cover outside of woodland is composed of small woods (non-NFI woods <0.1ha, 294.8kha), clusters of trees (<0.1 ha) and linear woody features (192.6 kha) and lone trees (78.2 kha).

The effect of tree planting on soil carbon is more variable, due to the potential for the loss of soil carbon during soil preparation and planting, which can result in net emissions for several years following establishment. The establishment of hedgerows and silvo-arable systems on arable land and soils with low carbon content have good evidence to support and increase in soil carbon stocks in the medium to long term, as it may take 50-100 years for soil carbon reach a new equilibrium (Falloon et al., 2004; Gregg et al., 2021; Upson et al., 2016). There is mixed evidence for the effect of tree planting on the carbon content of grassland soils (Gregg et al., 2021; Matthews, 2020; Shi et al., 2013), and planting trees in peat soils should be avoided in favour of peatland restoration where possible (see section 3.13). Linear woody features that act as wind breaks or occur along field contours can also reduce rates of soil erosion (Gregg et al., 2021).

Management and maintenance of linear and point woody features is important to preserving their potential value for biodiversity and function as boundary features, in addition to having a significant impact on their potential for carbon sequestration and storage. The 2007 Countryside Survey found that the total length of managed hedges in the UK had decreased from 1998, due to lack of management and degradation in to lines of trees, and that the overall diversity of all linear habitats had decreased (Norton et al., 2012). A comparison of the 2007 Countryside survey Data for Wales to that collected by the GMEP survey in 2016 found that the overall length of woody linear features had remained approximately constant, and a continuation of the trend reported by Norton et al. (2012) (Alison et al., 2020).

3.13.1.2 Co-Benefits and Trade-offs

No assessment.

3.13.1.3 Magnitude

Above ground carbon stocks in hedgerows reported in Gregg et al. (2021) vary between 26.7 t C ha $^{-1}$ and 131.5 t C ha^{-1}, across a variety of species compositions and managements in the UK, but are known to increase with hedgerow width and height.

Below ground carbon stocks may account for half of the overall carbon stock of hedgerows, with Axe et al. (2017) reporting that roots and below ground woody debris accounted for below ground stocks 38.2 tC ha^{-1} Rates of sequestration above and below ground are largely unknown, and most current estimates are from woodland systems.

A 30-40% increase in the extent of hedgerows estimated to lead to between a change in emissions of -1 and -4 Mt CO_2eq by 2050, assuming some management for biofuel production, when considered in concert with an increase in the area of agroforestry to cover between 5 and 15% of agricultural land (Thomson et al., 2020).

3.13.1.4 Timescale

>10 years
Rates of net carbon sequestration over the first 30 years will be largely dependent on the extent of initial soil carbon loss, species planted and management type, with the potential for planting to result in a net carbon sink or source in the first ten years (Prosser et al., 2022). Significant net carbon fixation is expected from tree planting on the time scale of 30-50 years (Gregg et al., 2021).

3.13.1.5 Spatial Issues

Likely to have successful broad implementation, with consideration given to appropriate soil type and species.

3.13.1.6 Displacement

Planting along boundaries and marginal habitats, or at low density is unlikely to result in significant displacement.

3.13.1.7 Maintenance and Longevity

Woody vegetation may require less maintenance in the form of fertilises and could interact with crops and grassland to reduce the need for pesticides. Carbon stocks in trees have a relatively high residency time (Matthews, 2020). However, preservation of standing carbon stocks active management will be necessary for the persistence of accumulated above and below ground carbon.

Hedgerows require regular maintenance to maintain their structure, and after approximately 40 years without structural maintenance, will lose their structural density, likely resulting in a loss of carbon stocks. Shorter term management and trimming is usually carried out on a scale of 1-3 years (Gregg et al., 2021).

3.13.1.8 Climate Adaptation or Mitigation

Carbon sequestered in woody features constitutes climate change mitigation.
Approaches to woody feature establishment could also be used which would result in woodlands that are more resilient to climate change.

3.13.1.9 Climate Factors / Constraints

Overall effect of climate change on the productivity of woody vegetation are unknown, with responses expected to vary significantly across species. Considering vulnerability to changing climatic conditions to maximise resilience when selecting species to plant may help mitigate risk from climate change.

3.13.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.13.1.11 Uptake

No assessment.

3.13.1.12 Other Notes

None

3.14 LITTER AND PLASTIC WASTE

3.14.1 ECPW-281: Use of biodegradable silage, crop cover mulches and planting trays to meet recognised compostable standard EN17033

Duplicated evidence base:
Use alternatives to fossil-based plastic mulches, such as green mulches or other biodegradable materials. Straw, shredded wood and other natural products can also be used as mulch [**ECPW-280**]

Co-Benefits and Trade-offs only

3.14.1.2 Co-Benefits and Trade-offs

The impacts of biodegradable plastic substitutes, including mulches, on carbon cycling are not yet well understood. The substitution of traditional fossil fuel derived products with biomass derived products would constitute emissions abatement. However, compostable standard EN17033 requires the ≥90% conversion of mulch's carbon into CO2 within 2 years under ambient soil conditions, implying that a significant enhancement of stocks from the addition of carbon rich material in this way is unlikely (Douglas G. Hayes & Markus Flury, 2018). Traditional mulches have been associated with lower rates of soil carbon accumulation than controls, whereas one study of biodegradable mulches showed no effect on rates of soil carbon accumulation relative to control (English, 2019).

The degradation of C rich materials has the potential to enrich soil carbon stock, or may have a priming effect on rates of soil respiration (Sayer et al., 2011). Analysis by English (2019) found the carbon content supplied by the degradation of C rich mulches was marginal compared to crop residues. The effect of soil inputs on the microbiome and microenvironment of soil may have consequences for carbon cycling in the long term, but estimates of effect size or direction are lacking (Serrano-Ruiz et al., 2021).

Food and fibre production	Area under production or yield and outside of ELM	N
Global, regional & local climate regulation	Above ground carbon sequestration	N
	Below ground carbon sequestration	LD*

[TOCB Report-3-5D Systems **ECPW-280**] There have been no direct tests of effects of plastic mulch removal on biodiversity, but one study in Poland showed that mulching with plastic foil had negative effects on the number of species and abundance of farmland birds at local and landscape scales (Skorka et al. 2013). Mulching with plastic foil had a negative effect on potential resources for birds including adult butterflies and their larvae and weed species.

3.15 MONITOR, PLANS, DATABASES, CONSULTATION AND RESULTING ACTION

3.15.1 ECCM-058: Monitor health of trees

3.15.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**T**
	Below ground carbon sequestration	**T**

There is good evidence that tree health is associated with rates of carbon sequestration, and tree mortality can deplete carbons stocks. Whilst the impact of health monitoring on sequestration rates has not been studied directly, there is a strong logic chain indicating that the early identification of damage and disease will enable responses to minimise the extent and severity of issues effecting tree health, with positive effects on carbon sequestration and storage potential by woodlands.

Carbon sequestration provided by trees are woodlands is expected to significantly contribute towards the achievement of net zero by 2050 in the UK (Gregg et al., 2021). Preventing carbon emissions due to the loss of existing carbon stocks in woodland is also important for this goal (Matthews, 2020). There is good evidence that poor tree health leads to reduced rates of carbon sequestration and storage (Boyd et al., 2013), in addition to significant economic cost. The impact of ash dieback alone has been estimated to the cost of £15 billion in Britain, approximately £4000 million of which is due to the loss of woodland ecosystem services (Hill et al., 2019). In the US, it has been estimated that tree disease is correlated with a reduction of above ground biomass accumulation by 9.33 Tg C yr^{-1} due to insect outbreak and 3.49 Tg yr^{-1} due to disease, compared to accumulation rates in trees not recorded as disturbed, controlling for variation in eco-province and carbon prior to any disturbance (Quirion et al., 2021). However the impact in the UK has not been quantified on this scale, but unfavourable conditions are likely to be found in a large percentage of the UK's woodland. ONS (2020) reported that an investigation done by NFI showed that 40% of woodlands are classified as unfavourable using an indicator for woodland condition based on grazing and herbivore damage using National Forest Inventory data calculated by Forest Research.

Local monitoring for tree health allows early intervention to occur to treat issues and prevent spread, and can subsequently inform National monitoring and research. Existing mechanisms allow for a Statutory Plant Health Notice (SPHN) to be issued following the identification of a diseased tree, to prevent spread and minimise damage by ensuring the felling of affected areas. Data reported by Forest Research (Statutory Plant Health Notices - Forest Research) show that 3163 SPHNs were issued in the UK from 2010 to 2019, of which 1124 were issued in England covering 4.5 kha.

3.15.1.2 Co-Benefits and Trade-offs

No assessment.

3.15.1.3 Magnitude

The current disease/pest burden on C sequestration in the UK is unknown, and future burden could be highly variable. In the US, it has been estimated that tree disease is correlated with a reduction of above ground biomass accumulation by 9.33 Tg C yr^{-1} due to insect outbreak and 3.49 TgC yr^{-1} due to disease, compared to accumulation rates in trees not recorded as disturbed, controlling for variation in eco-province and carbon prior to any disturbance (Quirion et al., 2021). On a per area basis, live trees with no recent record of disturbance sequestered on average 1.44 MgC ha^{-1} yr^{-1}, whilst live trees with evidence on insect disturbance sequestered 0.45 MgC ha^{-1} yr^{-1} and live trees with evidence of disease sequestered 1.04 MgC ha^{-1} yr^{-1}. Monitoring would also then need to be followed by appropriate management where possible, and it has no direct effects.

3.15.1.4 Timescale

<5 years for potential benefits, but the value of monitoring is likely to increase the longer it is carried out.

3.15.1.5 Spatial Issues

Broadly applicable.

3.15.1.6 Displacement

No assessment.

3.15.1.7 Maintenance and Longevity

Monitoring is required long-term to be effective and may be increasingly important as climate change increases the risk of emergent pests and disease effecting UK woodland.

3.15.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation. In this case, carbon emissions following tree death could be prevented and the capacity for sequestration maintained, where it would have otherwise decreased.

3.15.1.9 Climate Factors / Constraints

No assessment.

3.15.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.15.1.11 Uptake

Various resources exist to allow the reporting of pest and disease affecting UK trees, including the sources below:

TreeAlert - Forest Research[1]
Observatree - Pests and Diseases - Woodland Trust[2]
Protecting tree health in England | Forestry England[3]

3.15.1.12 Other Notes

n/a

3.16 NATURAL REGENERATION

3.16.1 ECCM-049: Create woodland through natural regeneration

3.16.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**
Global, regional & local climate regulation	Above ground carbon sequestration	LT***

[1] https://www.forestresearch.gov.uk/tools-and-resources/fthr/tree-alert/
[2] https://www.woodlandtrust.org.uk/trees-woods-and-wildlife/tree-pests-and-diseases/observatree/
[3] https://www.forestryengland.uk/blog/protecting-tree-health-england

	Below ground carbon sequestration	LT***

Woodland creation has the largest potential for increasing rates of carbon sequestration, out of any semi-natural habitat (Gregg et al., 2021). There is strong evidence that natural regeneration can result in the net sequestration of carbon, both above and below ground (Matthews et al., in prep.; Poulton et al., 2003). However, the likely variability in rates of carbon stock development across sites, and the likelihood successful establishment are not well known at a national scale.

Allowing natural regeneration to occur would result in the establishment of woodland and associated carbon stocks (see Table 1) without disturbing soil carbon. Soil loss due to site preparation and during the tree establishment process can be significant however, and potentially prevent net carbon sequestration from occurring for decades in some cases (Prosser, et al., 2022). A case study reported on by Matthews et al. (in prep.), where forest was planted on soils with high organic matter content, achieved net annual soil sequestration by around year 20, at which point 19.7 tonnes per hectare of soil carbon had been lost. Successful establishment also requires the presence of a suitable seed source, protection from browsing and grazing and the absence of undesirable plant species that could prevent successful establishment (Gregg et al., 2021).

Few studies of natural woodland regeneration have been carried out in England, although the process has been well studied elsewhere. A study of natural forest regeneration on abandoned agricultural land in England found that 86% woody vegetation cover was achieved after 23 years at one site (2.1ha), and 100% cover was achieved after 53 years at a second site (3.9ha) (Broughton et al., 2021). Colonisation was spatially clustered and differed from ancient woodland in terms of species composition. A global analysis by [Cook-Patton 2020] found plant carbon stocks in naturally regenerating temperate broadleaf forests approximating 75 tC ha^{-1} by 100 years of age. Matthews et al. (in prep.) reported on two studies of natural woodland regeneration in South East England (mixed broadleaf woodlands, Broadbalk and Geescroft), originally reported by Poulton et al. (2003). The sites were previously used for arable crop production. Both sites showed a comparable increase in above ground carbon stocks to planted woodland after 45 years, with the exception of fast growing Sitka spruce which had higher carbon stocks. Broadbalk showed the largest increase in soil carbon stocks after 45 years of all sites, and averaged double the rate of carbon sequestration in Geescroft between ages 66 and 85, and 85 and 100. Although early rates of sequestration (0-25 years) were close to half those observed in planted woodland, in both naturally regenerating forests, there is a suggestion that rates of carbon sequestration may be peaking later than in planted forests, with both naturally regenerating woodlands showing higher rates of sequestration than planted woodland (except sitka spruce) from 66-85 years (Matthews et al., in prep.). Despite the extensive information available for these two sites, the timing and rate of carbon sequestration that can be achieved through natural abandonment is likely to be highly sensitive to the environment and baseline conditions, and as such extrapolating to a national scale should be approached with caution. Expert opinion suggests that the examples presented in Mathews (in prep) and Poulton et al. (2003) are likely sites with particularly good potential for natural regeneration. It is also possible for natural regeneration to fail to occur entirely (Demeter et al., 2021; Holl et al., 2018).

Assisted natural regeneration may also be a viable option in parts of the UK (M. C. Evans et al., 2015; Yang et al., 2018), although the evidence base for assisted natural regeneration is relatively undeveloped. Although the requirement for hands on management will reduce the economic savings that can potentially be achieved through natural regeneration alone (O'Neill et al., 2020), managing community composition at early stages of succession may promote more rapid establishment and increase the likelihood of the resultant forest being the desired composition and having higher value for biodiversity (Bowen et al., 2007).

3.16.1.2 Co-Benefits and Trade-offs

No assessment.

3.16.1.3 Magnitude

The maximum potential carbon uptake from woodland creation in the UK is site dependent. The largest carbon storage possible over 50 years is estimated to be by Sitka Spruce in the North and South-West of England, approaching 1450 tCO_2 ha^{-1}. However, aspen and sycamore are expected to provide higher rates of sequestration in much of the midlands and East of the country (reported by Gregg et al., 2021, using data from the Forestry Commission and analysed by Forest Research).

Using data from the woodland carbon code, Gregg et al. (2021) estimate sequestration rates for un-thinned, mixed, native broadleaved woodland as shown in Table 1. Over a 30 year period, they also report representative values for sequestration by beech (-11.5 t CO_2eq ha^{-1} yr^{-1}) and oak (-15.5 t CO_2eq ha^{-1} yr^{-1}) but state that woodlands of beech or oak are likely to be less resilient to climate change.

Matthews et al. (in prep.) presents and analysis of carbon sequestration and storage over time as a result of afforestation in nine study sites with varied compositions and managements. The estimated rate of carbon sequestration in living woody biomass during a 100 year period from the time of woodland creation was in the range -1 to -3 tC ha $^{-1}$ y $^{-1-}$ (-3.7 to -11 tCO_2 ha $^{-1}$ y $^{-1}$), and commonly takes a value around -1.4 tC ha $^{-1}$ y $^{-1}$ (-5 tCO_2 ha $^{-1}$ y $^{-1}$). This summary excluded stands managed for clear felling.

Matthews et al. (in prep.) also report model estimates for 11 contrasting options for woodland creation involving tree species mixtures and management practices relevant for the UK. These options include natural recolonisation of non-wooded land with broadleaf trees and 'light' subsequent management, examples of mainly moderate- and fast-growing commercial coniferous woodlands, and managed woodlands composed of complex tree species mixtures. The mitigation contributed by the woodland options contrasts in magnitude over time, with different options providing the most mitigation benefits at different times (between 2022 and 2100 and beyond) and in different ways (involving direct carbon sequestration or GHG emissions savings through provision of wood products, to varying extents). It should be noted that the different woodland options are not always interchangeable on the same site or in the same location within the UK. Planting 1 hectare of woodland in 2022 is estimated to result in net carbon sequestration in woodlands and wood products over the period from 2022 to 2050 of between 0.8 and 13.8 tCO_2 ha $^{-1}$ y $^{-1}$, with a mean estimate for all 11 options of 5.7 tCO_2 ha $^{-1}$ y $^{-1}$. If GHG emissions savings arising from utilisation of additional wood products and wood fuel are allowed for, these estimates increase to between 1.7 to 32.0 tCO_2 ha $^{-1}$ y $^{-1}$, with a mean estimate for the 11 options of 12.4 tCO_2 ha $^{-1}$ y $^{-1}$.

3.16.1.4 Timescale

>10 years
Small initial carbon gains could be seen in the first 10 years, however the majority of sequestration will happen after that point. For rates of sequestration across all woodland carbon pools assessed by Matthews et al., in prep.

3.16.1.5 Spatial Issues

Targeted.

3.16.1.6 Displacement

There is a strong likelihood of displacing agricultural activity as a result of natural regeneration.

3.16.1.7 Maintenance and Longevity

In the long term, protection of regenerating forests will be required to ensure their conservation. Management may also be required long-term to maintain desired conditions and species compositions.

3.16.1.8 Climate Adaptation or Mitigation

Carbon sequestration has the potential to promote climate change mitigation and provide microclimate buffering.

3.16.1.9 Climate Factors / Constraints

Naturally regenerating woodland could be vulnerable to climate change, along with their carbon stocks.

3.16.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.16.1.11 Uptake

No assessment.

3.16.1.12 Other Notes

None

3.16.2 ETPW-171: Allow natural regeneration and extension of existing habitat (e.g. hedgerows, scrub, rough grassland)

3.16.2.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**LT***
	Below ground carbon sequestration	**LT***

Extension of existing habitat – more likely due to presence of a seed source but it depends on the history of the site and the starting conditions. Protection from grazing pressure early on can be important, particularly for woody vegetation.

The importance of habitat extension through for carbon will be determined by the habitat being established, see actions under create hedgerows, scrub and grassland for more information. However, this is conditional on the success of natural regeneration. Natural regeneration has the potential to prevent the carbon loss from soil, often associated with land cover change (Gregg et al., 2021) + Matthews et al., in prep.. However, resulting increases in carbon sequestration and storage may be slower to occur (Matthews et al., in prep.), and community composition may differ from expectation without intervention. Natural regeneration will not negate the need for long term management in some cases.

In general, the maintenance of hedgerow structure is highly dependent on management on decadal time scales, without which structures degrade and lose structural density (Emmett et al., 2010; Norton et al., 2012).

The propensity of scrubland to colonise new areas unassisted is highly variable across species, some of which require active maintenance and others of which are typically actively suppressed to maintain their extent. Therefore, the success form the natural regeneration of scrub is also likely to be highly variable.

There is some evidence that the expansion of grassland habitat by natural regeneration is slower than alternative methods of establishment (seed sowing and hay transfer), but also that species richness is more likely to be maintained over successive years following natural regeneration (Rydgren et al., 2010). Auestad et al. (2015) found that all grassland restoration treatments (natural regeneration on bare soil, seeding bare soil, and hay transfer from local or distant donors) found that all methods converged in terms of species richness and composition after 2 years, which reflected that of the local donor site, suggesting that natural expansion of grassland may be successful. Changes in carbon content were not reported by either study.

The natural mature vegetation community following succession in much of the UK is woodland, with the exception of a few sites limited by climate (Gregg et al., 2021). As such, to maintain a non-woodland community, management intervention is likely to be required in the long term.

Naturally regenerating communities are likely to be highly influenced by available seed sources (Auestad et al., 2015; Gregg et al., 2021), and therefor the potential for selection stands that may be more resilient to climate change will not be possible.

3.16.2.2 Co-Benefits and Trade-offs

No assessment.

3.16.2.3 Magnitude

No evidence was found for the magnitude of carbon that could be sequestered by allowing the extension of grassland, scrubland and hedgerow habitats by natural regeneration, specifically. Estimates for the carbon sequestration or these habitats when actively established can be found in section 3.5.

3.16.2.4 Timescale

Unknown, see magnitude.

3.16.2.5 Spatial Issues

The success of natural regeneration, particularly when achieving the conditions in a reference habitat are desired, is likely to be highly variable. Sites will likely need targeting in areas with appropriate conditions and seed sources.

3.16.2.6 Displacement

Displacement of agricultural activity is possible from this action, although less so from hedgerow expansion.

3.16.2.7 Maintenance and Longevity

Naturally regenerating habitats may require management to maintain a desirable condition and prevent further succession.

3.16.2.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.16.2.9 Climate Factors / Constraints

Naturally regenerating habitats could be vulnerable to climate change, along with their carbon stocks.

3.16.2.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.16.2.11 Uptake

No assessment.

3.16.2.12 Other Notes

None

3.17 RESTORATION, MANAGEMENT AND ENHANCEMENT – COASTAL

3.17.1 EHAZ-074: Active management of coastal realignment

*Duplicated evidence base: **EHAZ-089** Restore/ manage natural water flow in coastal habitats*

3.17.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	*
Global, regional & local climate regulation	Above ground carbon sequestration	**LT***
	Below ground carbon sequestration	**LT****

Actively managed coastal realignment involves the controlled weakening and removal of sea defences to allow coastal forces to modify formerly terrestrial habitats. Depending on the land height relative to the sea, a variety of coastal wetland habitats can be created in this way (RSPB, 2021). In most cases this process will create saltmarsh, sand flats, mud flat and shingle, which can have a role as soft coastal defences. Managed realignment can be highly controversial with landowners and managers who lose land to inundation or tidal processes. For the most benefit for carbon sequestration, managed realignment should be targeted at coastal arable land. Coastal squeeze is a significant threat to habitats that have been identified as significant stores of blue carbon, and which have historically been converted to alternate land uses or degraded (Burden et al., 2019).

Coastal habitats have a varied potential for below ground carbon sequestration, which in the case of salt marshes does not show signs of saturation in in the same way as soil and woodland carbon (Burden et al., 2019; Element Energy & UKCEH, 2021). However, the capacity for above ground carbon sequestration is relatively limited. For a more detailed account of the carbon sequestration potential of coastal habitat creation via managed realignment, see section 3.7.

It may also be worth considering the impact of alternatives to managed realignment on carbon sequestration and storage. The continuation of sea level rise without managed realignment will result in the net loss of coastal habitats and their future sequestration potential, although carbon stocks may be preserved to some degree in marine sediments (Burden et al., 2020; Gregg et al., 2021; Parker et al., 2021). An alternative to managed coastal realignment, that maintains or increasing carbon sequestration potential may be to convert land that would be affected by managed realignment to another land cover capable of higher rates of carbon sequestration than coastal habitats can provide in a short to medium scale time frame (for example, forestry managed for carbon sequestration, see section 4.7.4). However, due to continued sea level rise and risk of extreme climatic events, there is a possibility the hard coastal defences in some parts of the coastline will fail in an uncontrolled fashion. This can result in significant damage if the capacity of associated estuaries and floodplains is rapidly exceeded (Wadey et al., 2013). In addition to the risk this entails for human life, property and services, such an event could result in significant ecological and environmental damage.

When considered, expert opinion suggest that managed realignment should be considered as part of the wider coastal management plan, to ensure the conservation of coastal habitats at the national scale and local economic, cultural and ecological factors.

3.17.1.2 Co-Benefits and Trade-offs

Coastal flood defence.

3.17.1.3 Magnitude

See section 3.7 for estimates of carbon accumulations rates and stocks that can be achieved via coastal habitat creation and from managed realignment on a habitat basis.

3.17.1.4 Timescale

See section 3.7

3.17.1.5 Spatial Issues

See section 3.7

3.17.1.6 Displacement

There may be displacement of economic activities by managed realignment as land is potentially taken out of production.

3.17.1.7 Maintenance and Longevity

Where secondary coastal defence or tidal regulation is required as part of managed realignment, ongoing management will be necessary.

3.17.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.17.1.9 Climate Factors / Constraints

Continued sea level rise will likely result in the loss of existing coastal habitats to marine ecosystems, as a component of coastal squeeze. Carbon that has been historically sequestered by coastal habitats may be vulnerable to disturbance by wave action, however, once no longer subject to tidal action and wave force the carbon is likely to be relatively stable in sub tidal sediments (Parker et al., 2021) .

Burden et al. (2019) argue that once established, many coastal ecosystems have the capacity to self-regulate and respond to rising sea levels by migrating inland with sea level rise, in the absence of hard coastal boundaries and subsequent coastal squeeze.

3.17.1.10 Benefits and Trade-offs to Farmer/Land-manager

Land managers and farmers face the loss of potentially productive land to managed realignment.

3.17.1.11 Uptake

Government policy for shoreline management have been published, which identify approaches to managing the flood and coastal erosion risks to the coastline[4].

Managed realignment can be highly controversial with landowners and managers who lose land to inundation or tidal processes and can be associated with significant up front capital costs, which may deter uptake (Element Energy & UKCEH, 2021).

3.17.1.12 Other Notes

None

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[4] https://www.gov.uk/government/publications/shoreline-management-plans-smps

3.17.2 **EHAZ-070EM: Enhance / maintain sand dunes**

Duplicated: **ECCM-046:** Use controlled grazing on intertidal, saline, salt marsh and coastal grassland habitats & **ECPW-083:** Control grazing on sand dunes.

3.17.2.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**LD***
	Below ground carbon sequestration	**LD***

Managements of sand dunes in the UK is primarily aimed toward the conservation of sand dune systems, the reversal of over-stabilisation, restoring degraded systems as a result of grazing and disturbance levels, and creation of diverse sand dune habitats for biodiversity (Jones et al., 2021). Whilst sand dunes can store significant carbon stocks in sediments, particularly in dune grassland, these habitats store more carbon as dunes become more stable, and drier (Beaumont et al., 2014; Gregg et al., 2021; Jones et al., 2008). This means that managements that are necessary for sand dune conservation may have a negative effect on potential carbon stocks, although management is likely to have limited significant to national carbon sequestration potential. Given that our understanding of carbon dynamics in coastal dunes is based on a relatively small number of studies, making generalised inferences about the effect of management on dune carbon stocks should be done with caution.

<u>Vegetation management</u>
Vast majority of dunes in the UK are considered overgrown, with low rates of natural dune processes. In particular, early dune specialists are largely dependent on targeted management. Over-stabilisation is counteracted by grazing pressure, localised turf stripping and invasive species management where relevant (Jones et al., 2021). Grazing pressure has been found to increase species diversity but reduce the coverage of tall grass species in dry dunes (Plassmann et al., 2010). Plant species diversity has been correlated with higher rates of carbon sequestration in other habitats (Alison et al., 2019; Chamagne et al., 2016; Jucker et al., 2015; Kelty, 2006), but we know of no evidence for diversity impacting carbon sequestration in dune systems. (Ford et al., 2012) studied the effect of grazing management on the carbon stocks of coastal dunes in NW Wales, with a history of livestock grazing. Exclusion zones were maintained for 6 years, but no significant difference in overall carbon stock was found between site with no grazing, rabbit grazing and livestock grazing. This was despite a significant increase in vegetation height and litter in the no grazing treatment, relative to both grazing treatments, potentially due to the relatively small contribution of vegetation to sand dune carbon stocks (Gregg et al., 2021).

<u>Hydrology</u>
Hydrology is an important driver of sand dune dynamics. many sand dune slacks have been affected by a lowering of the water table due to abstraction or drainage, which can result in their colonisation by vegetation that is typical of drier dune habitats and the loss of dune slacks (Jones et al., 2021). There is evidence that rates of soil formation are higher in wet dune habitats (Jones et al., 2008), although the soil carbon stocks of dune slacks were significantly higher in dune grasslands (Beaumont et al., 2014). Jones et al. (2021) show that water use by forestry can significantly affect the water table in dunes. They show that the clear felling of forest would positively impact hydrological conditions for nearby dune conservation and that cycles of clear felling and thinning have likely been beneficial to dunes in the past. There is strong evidence that woodland and forestry can contribute significantly more carbon sequestration and storage the coastal dune systems however, and as such removing woody vegetation to benefit nearby coastal dunes is very likely to result in net carbon emissions.

3.17.2.2 Co-Benefits and Trade-offs

No assessment.

3.17.2.3 Magnitude

Ford et al. (2012) studied the effect of grazing management on the carbon stocks of coastal dunes in NW Wales, with a history of livestock grazing. Exclusion zones were maintained for 6 years, but no significant difference in overall carbon stock was found between site with no grazing, rabbit grazing and livestock grazing.

The effects of other managements of sand dune carbon stocks are unknown. Baseline carbon stocks are highest in the sediments of dune grasslands (178.7 tC), followed by dune slack (46 tC) and were negligible for more mobile dunes (Beaumont et al., 2014). Vegetation biomass (above and below ground) was also highest in dune grassland (93.7 tC) but was lowest in dune slacks (19 tC). Mobile dunes were estimated to contain 67.5 tC in vegetation biomass. Average storage was estimated by Gregg et al. (2021) as 0.0095 tC ha^{-1} in dune sediments to a depth of 15cm and 0.005 tC ha^{-1} in dune vegetation, although estimates are associated with a low degree of confidence.

Jones et al. (2008) report average sequestration rates of -0.582 t C ha^{-1} y^{-1} for dry dunes and -0.73 t C ha^{-1} y^{-1} for dune slacks, resulting in an average rate of -0.595 t C ha^{-1} yr^{-1} estimated from a chronosequence of dune development in North Wales spanning 140 years.

3.17.2.4 Timescale

>10 years. Given work by (Ford et al., 2012), it is likely that any significant changes in soil carbon stocks due to grazing pressure will be slow to develop.

3.17.2.5 Spatial Issues

The principles behind the management vegetation and sand dune hydrology are broadly applicable, although the implementation of management should be site specific.

3.17.2.6 Displacement

No assessment.

3.17.2.7 Maintenance and Longevity

In the absence of management, it is likely that natural dune succession will resume, in the absence of site degradation.

3.17.2.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.17.2.9 Climate Factors / Constraints

No assessment.

3.17.2.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.17.2.11 Uptake

No assessment.

3.17.2.12 Other Notes

In light of the relatively small carbon stocks in sand dunes within England, managements in the system will not have a substantial impact on national carbon sequestration rates.

3.17.3 ETPW-180EM: Enhance/manage inter-tidal and saline habitats*

3.17.3.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**N**
	Below ground carbon sequestration	**L***

Management options for intertidal sediments are limited, and the authors know of no studies addressing this question. Based on expert opinion, the conservation of carbon stored in intertidal sediments is most likely to be impact by disturbance, and therefore minimising sources of sediment disturbance in coastal mud flats may be beneficial. The magnitude of this problem is thus far unknown, and prevention of disturbance may conflict with recreational and economic use of coastal waters. Research into the importance of intertidal and marine sediments as carbon stores and their responses to disturbance is required to fill this evidence gap.

3.17.3.2 Co-Benefits and Trade-offs

No assessment.

3.17.3.3 Magnitude

Unknown.

3.17.3.4 Timescale

Unknown.

3.17.3.5 Spatial Issues

Unknown.

3.17.3.6 Displacement

If the regulation of disturbance involves reducing tourist or economic activity, these could be displaced to other habitats.

3.17.3.7 Maintenance and Longevity

No assessment.

3.17.3.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.17.3.9 Climate Factors / Constraints

No assessment.

3.17.3.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.17.3.11 Uptake

No assessment.

3.17.3.12 Other Notes

None

3.17.4 ETPW-081EMX: Enhance manage salt marsh

3.17.4.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	*
Global, regional & local climate regulation	Above ground carbon sequestration	**L***
	Below ground carbon sequestration	**L***

There are few studies assessing the impact of salt marsh management on carbon sequestration rates and stocks. Although logic chains support the principle that reduced grazing pressure should result in an increase in above, and subsequently below ground carbon stocks, grazing was found to have no significant effect on below ground carbon stocks across 22 salt marshes in the UK, by (Harvey et al., 2019).

Above ground biomass was significantly affected, with lower grazing intensity leading to more biomass, and intermediate grazing leading to the highest species richness. Above ground biomass with no grazing was found to be highly variable, and was associated with a larger sample size than other stocking densities. (Davidson et al., 2017) also found that grazing had no significant effect on soil carbon in European salt marshes in a meta-analysis of 75 studies. A global meta-analysis of the effects of coastal managements on carbon balance found that sediment manipulations had a negative effect of carbon stocks in natural and stored sites, although response metrics varied between studies (O’Connor et al., 2020). Restored sites where hydrology had been manipulated (water tables restored) high higher carbon stocks than degraded sites but not natural salt marshes. The majority of managements had no significant impact on carbon stocks.

Overall, in the absence of significant overgrazing and degradation, the limited available evidence suggest that management and enhancement of saltmarsh is unlikely to have a substantial impact on carbon sequestration potential, although logic chains suggest it is a possibility.

3.17.4.2 Co-Benefits and Trade-offs

No assessment.

3.17.4.3 Magnitude

Although logic chains support the principle that reduced grazing pressure should result in an increase in above, and subsequently below ground carbon stocks, grazing was found to have no significant effect on below ground carbon stocks across 22 salt marshes in the UK, by (Harvey et al., 2019). Above ground biomass was significantly affected, with lower grazing intensity leading to more biomass, but the magnitude of the effect was small, with mean difference in above ground biomass between a stocking density 0 LUS ha^{-1} yr^{-1} and 3.5 LUS ha^{-1} yr^{-1} reported as approximately 0.1 g cm^{-2}, or 0.001 t ha^{-1}.

3.17.4.4 Timescale

Unknown.

3.17.4.5 Spatial Issues

Unknown.

3.17.4.6 Displacement

Reducing grazing pressure in salt marsh may cause agricultural displacement, however the risk of this is low.

3.17.4.7 Maintenance and Longevity

No assessment.

3.17.4.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.17.4.9 Climate Factors / Constraints

No assessment.

3.17.4.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.17.4.11 Uptake

No assessment.

3.17.4.12 Other Notes

n/a

3.18 Restoration, management and enhancement – Cropland

For inputs, tillage etc. see soil management and protection

3.18.1 ECCM-021: Farm perennial crops

Duplicate evidence base:
Diversify arable rotations (including cover and catch crops, over and under sowing). [**ECCM-001**];
Use crops / varieties with differing root structures to improve soil structure and nutrient uptake [**ECPW-265**]

3.18.1.1 Causality

Energy use/ use of renewables		**T***
Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**T***
	Below ground carbon sequestration	*****

Perennial crops increase root carbon input and reduce soil disturbance thus potentially increasing SOC (Rollett & Williams, 2020). Much of the work investigating the management of perennial arable crops has been carried out in temperate regions which is relevant for UK agriculture. Integrating grasses into crop rotations can enhance carbon inputs to soil and reduce SOC losses via decomposition, resulting in carbon sequestration (Conant et al., 2017). Even small-scale grassland creation can profoundly affect SOC sequestration. According to one study, 0.1 to 2.4% of 1990 UK CO_2-C emissions could be sequestered using grass margins on arable fields (Falloon et al., 2004). A more recent study demonstrated significant increases in soil C stocks following land use change from arable to perennial SRC willow coppice (Rowe et al., 2016).

King and Blesh (2018) compiled a database of 169 cropping systems, which categorised rotations into three broad groups: 1) grain only, 2) grain with cover crops and 3) grain with perennial crops. According to their meta-analysis, crop rotations that included a perennial crop stored 12.5% more SOC than grain-only rotations (King & Blesh, 2018). A more recent global meta-analysis which included annual and perennial crops reported that maize and ryegrass had the greatest allocation to the soil of 1.0 t C ha^{-1} yr^{-1} (Mathew et al., 2020). These meta-analyses reinforce the suggestion of an earlier study that converting from annual to perennial crops could potentially accumulate up to 1.7 t C ha^{-1} yr^{-1} (Crews & Rumsey, 2017).

Much of the carbon lost from arable soils is lost via erosion (Gregg et al., 2021; Owens et al., 2006). Providing year-round vegetative cover through the use of perennial crops is considered the most effective measures to mitigate carbon losses via erosion (Gregg et al., 2021). In addition, perennial crops could be barriers to sediment transport in the wider landscape. Land set aside for perennial crops could also yield biodiversity benefits for some plants and invertebrates (Alanen et al., 2011), though Freibauer et al. (2004) caution that set-aside can cause issues with weeds in future years, and that switching to perennial crops leaves less room to respond to short- term market changes. Perennial bioenergy crops can also be cultivated, and are reviewed in section 3.4.3, however this would increase the risk of agricultural displacement.

3.18.1.2 Co-Benefits and Trade-offs

No assessment.

3.18.1.3 Magnitude

One study suggests that perennial crops and grasses store up to 0.62 t C ha^{-1} yr^{-1} (Dawson & Smith, 2007). Another study suggested that the most effective crop rotations for SOC storage were those that included a perennial crop, which increased SOC concentrations by Freibauer et al. (2004) report that permanent revegetation of arable land e.g. introducing perennial components, short rotation coppicing or perennial grasses, could increase SOC by 0.5–1.9 t C ha^{-1} yr^{-1}. Between 0.1 and 2.4 % of 1990 UK CO_2-Carbon emissions could be sequestered using grass margins on arable fields (Falloon et al., 2004). It has been argued that 7.7 % increases in topsoil (0-20 cm) SOC in Sweden were attributable to increases in grass ley as a fraction of total agricultural area (Poeplau et al., 2015). Elsewhere, experimental data on SOC collated from EU countries revealed that ley-arable farming or afforestation of arable land could sequester more carbon than addition of organic manures (Smith et al., 1997).

3.18.1.4 Timescale

0-5 years

Many of the studies reviewed by Alison et al. (2019), Rollett and Williams (2020), and Matthew (2020) cover periods of about 5 years.

3.18.1.5 Spatial Issues

The potential for SOC accumulation may be greatest where there is already a low SOC stock (Powlson et al., 2011). For example, one study found the sites with the lowest arable SOC stocks to begin with experienced the greatest increases in SOC storage following conversion to short rotation willow coppice (Rowe et al., 2016).

3.18.1.6 Displacement

There are several possible approaches to integrating perennial crops or more permanent vegetation cover into arable rotations (see Uptake) with varied levels of supporting evidence for impacts on soil carbon and feasibility, which each will differently impact displacement. Although converting arable land to perennial crops or set-aside will likely yield significant SOC storage benefits, there are highly likely to result in a corresponding reduction in food production (Moxley et al., 2014). Alison et al. (2019) has argued that this could mean that food production may simply occur elsewhere (including international displacement) with emissions that offset local increases in SOC. Converting land used for arable food production to bioenergy crop production will displace food production.

3.18.1.7 Maintenance and Longevity

The inclusion of 3-year grass or grass-clover leys in a 5-year arable rotation may also have positive effects that saturate after about 30 years (Johnston et al., 2017). Furthermore, a fraction of SOC that accumulates will stabilise with an expected half-life on the order of decades to centuries (Powlson et al., 2011).

3.18.1.8 Climate Adaptation or Mitigation

The significant positive carbon sequestration effect of this intervention, coupled with the lack of issues associated with either N_2O or CH_4 emissions, should contribute to climate mitigation overall. The effect is likely to be most pronounced with a conversion of arable land to grassland (Smith et al., 2008).

3.18.1.9 Climate Factors / Constraints

Matthews et al. (2020) reported that higher SOC accumulation was recorded under clayey soil and warmer climates across all crop types analysed. Another study suggested that more positive effects of set-aside and land use change occur in moist rather than dry climates (Smith et al., 2008).

3.18.1.10 Benefits and Trade-offs to Farmer/Land-manager

If erosion rates and nutrient leaching are reduced, and more SOM accumulates to improve soil structure, it could improve crop yields. On the other hand, if the trade-off of reduced food production highlighted by Moxley et al. (2014) does occur, that would reduce the profitability of the land. This issue would likely be compounded if issues with weeds and short-term market changes (which farmers would be more vulnerable to) occur (Freibauer et al., 2004).

3.18.1.11 Uptake

There are several approaches that could be used to integrate perennial crops or more permanent vegetation cover to arable rotations, with varying barriers to uptake and different levels of supporting evidence (with respect to their impact on soil carbon).

Including perennial grasses in arable rotations is immediately implementable and will be associated with less displacement of crop production than full conversion to (for example) perennial biofuel crops, but will still impact food production and profitability. This practice will not confer protections to soil outside of the relevant rotation. Agroforestry has the potential to improve arable soil carbon stocks through perennial vegetation cover with displacing all food production, with some practical barriers to uptake, and for a full review of this practice see section 3.6.

Converting land used for arable food production to bioenergy crop production will displace food production and is reviewed at 3.4.4.

Additional opportunities also exist to develop a range of perennial crops through hybridisation, as numerous annual grain or row crops such as maize and oats have closely related wild perennial relatives (Cox et al., 2006). In the USA, the perennial wheat relative, intermediate wheatgrass (IWG) *Thinopyrum intermedium*, marketed as Kernza, is already being used in food products (Rollett & Williams, 2020). However, IWG has been shown to have a lower vegetative biomass and yield than annual wheat (Culman et al., 2013). These options are unlikely to be suitable for widescale application in the near future, but have potential in the longer term.

3.18.1.12 Other Notes

None

3.19 Restoration, management and enhancement – Grassland

Grassland is not a perpetual carbon sink, as a result of the saturation of soil carbon stocks and limited accumulation of carbon in above ground biomass. Management for carbon sequestration in grasslands should primarily be targeted at maintaining existing high carbon stocks (Smith, 2014). Additionally, changes to management that reduce on site and associated life cycle emissions are possible. Managements to increase the capacity for soil carbon storage in grassland include the integration of legumes and sward diversification (small effect) and the adoption of silvo-pasture practices (potentially a large effect). Any management that involves ploughing or soil disturbance is likely to have a negative short term impact on carbon stocks.

3.19.1 ETPW-104: Reduce stocking rate (grazing) to restore structure and flowering, maintain ground cover, and reduce poaching

Duplicated evidence base:

Reduce stocking density or remove livestock grazing where likely impacts on sensitive habitats and species (aquatic and terrestrial) [**ECAR-035**]; Use low intensity mixed livestock grazing [**ETPW-105**]; Use low-intensity grazing systems using biodiverse sward mixtures [**ECCM-014**]; **ECPW-181**: Conversion to a more extensive system including reversion from high risk forage to grass and whole crop and reduced inputs; **EHAZ-010Y** Enhance or manage permanent grasslands; **ETPW-106** Manage timing of grazing and select livestock type to allow flowering and seed return, and control competitive and invasive species.; **ETPW-150** Manage localised grazing pressure.

3.19.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	*****
	Below ground carbon sequestration	**TD***

The interactions between grazing and grassland carbon stocks are complex, with evidence suggesting both intensive grazing and no grazing can be detrimental for soil carbon stocks in some cases, and that responses to a change in management are site dependent (Abdalla et al., 2018). The majority of evidence suggests that low intensity grazing will be beneficial for grassland carbon stocks. However, the emissions from livestock themselves should also be considered.

Grassland carbon is primarily stored in the soil, and maintained by primarily by below ground litter (root systems) and exudates. Ward et al. (2016) estimate that 2097 Tg C is stored in grassland soils across Great Britain to a depth of 1m and with 60% of this below 30cm. The Countryside Survey 2007 found that improved grassland contained 67.2 t ha $^{-1}$, neutral grassland contained 68.6 t ha^{-1} and acid grassland contained 90.6 t ha $^{-1}$ (Emmett et al., 2010). Under constant management, carbon stocks in grassland will reach an equilibrium state, where carbon stocks are neither increasing nor decreasing (Smith, 2014). However, responses by soil carbon stocks to a change in management can take many years to stabilise.

Where nutrients are limiting vegetation growth, grazing can stimulate addition productivity due to the associated increase in nitrogen inputs (manure and urea) (Rollett & Williams, 2020). There is also some evidence that grazing can stimulate root exudation of carbon to soils, and may encourage below ground productivity (Alison et al., 2019). However, high stocking rates can reduce sequestration rates by removing above ground biomass, which may also reduce inputs from the vegetation to the soil (Rollett & Williams, 2020); Prosser et al. 2022). Over-grazing can result in the complete loss of vegetation cover in places, causing elevated carbon loss to erosion and soil compaction (Alison et al., 2019). Urine can lower soil pH which affects the biogeochemical processing of soil carbon and may lead to higher rates of soil carbon mobilisation for decomposition (Rollett & Williams, 2020). Grasses will also vary in their capacity to support grazing rate whilst maintaining root growth and exudate production, with C4 grasses likely to support higher grazing rates than C3 grasses (Prosser et al., 2022).

Multiple reviews of the effects of reduced grazing intensity have been conducted with the majority finding that extensive grazing systems are associated with higher soil carbon stocks or rates of sequestration than intensive grazing systems, and these reviews have been widely discussed (Alison et al., 2019; Gregg et al., 2021; Rollett & Williams, 2020). Eze et al. (2018) conducted a global meta- analysis of the effects of grazing intensity on soil carbon stocks and found a consistent and significant decrease in carbon stocks with increasing grazing pressure from no grazing to light (-6.9%), to moderate (-13.2%) and to heavy grazing (-27.1%). Their definition of levels of grazing intensity is not quantitative, and not all contributing studies may be informative to circumstances in the UK, and prior land use and state of degradation is not accounted for. (Conant et al., 2017) reviewed the impacts of improved grazing (which included lower stocking rates, removal of grazing livestock, rotational or short duration grazing and seasonal grazing) on soil carbon concentrations and associated rates of change. Improved grazing led to a slow increase in soil carbon stocks at 0.28 t $Cha^{-1}yr^{-1}$. Mean changes in soil carbon stocks were positive following a change to improved grazing management, despite a majority of studies showing a decrease in carbon stocks.

For studies that reported a change in carbon concentrations only, improved grazing resulted in a small increase in concentration from 2.62% to 2.89% on average (approximately a 10% increase of baseline), but 15 out of 40 studies reported a decrease in carbon concentration. The conversion to a more extensive grazing system was found to increase rates of soil carbon sequestration in Europe, including in the UK, using the model ORCHIDEE-GM to model net biome production. The effects of a move to more extensive livestock management, including reduced livestock numbers, in European grasslands between 1991 and 2010 was modelled by (Chang et al., 2016) using ORCHIDEE-GM, for sites including the British Isles. They estimated that the change to more extensive management had driven an increase in soil carbon and reduction in GHG emissions by a total of 1.2-1.5 Gt CO_2 eq over the study period, and were responsible for an increase in net biome productivity in the British Isles by approximately 1.5 gC m^{-1} yr^{-1}.

Reducing grazing pressure has been associated with a small increase in above ground carbon stocks, but the increase is unlikely to be nationally significant assuming the habitat is maintained as grassland as opposed to scrub or silvopasture (see reviews in sections 3.6-3.13). Over time, changes in above ground biomass will translate to below ground biomass which is associated with greater permanence. Grazing systems at a global aggregate scale currently emit more GHGs than they sequester in the form of SOC stocks. Although grazing-induced sequestration offers a climate mitigation strategy, it should be promoted by targeted farming strategies where permissible, and ultimately its global mitigation potential is lower than frequently suggested (Godde et al., 2020).

Overall, both expert opinion and existing evidence suggest that a move to more extensive grazing and reduced grazing pressure is likely to benefit soil carbon stocks, but information specific to the the UK is somewhat lacking and management decisions are likely best guided by site specific circumstances.

3.19.1.2 Co-Benefits and Trade-offs

[TOCB Report-3-5D Systems **ECPW-181**] This is a general nutrient management action, from which we expect an overall reduction in nutrient inputs, with various positive effects on biodiversity. At the whole-farm level, these could be quite significant. However, effects will depend on the context of baseline nutrient levels.

3.19.1.3 Magnitude

A reduction in grazing pressure has variable effects on net ecosystem carbon balance. Estimates of the effect that include reduced emissions from livestock directly suggest that reduced intensity of livestock management between 1991 and 2010 led to the sequestration of -1.5 gC m^{-2} yr^{-1} in the British Isles, separate from any associated changed in land cover (Chang et al., 2016).
Abdalla et al. (2018) summarise the general impacts on SOC under different grazing pressures and climate conditions:

- ***Dry warm climate***: SOC <u>declines</u> under all conditions apart from low grazing intensity where SOC <u>increases</u> by 5.8%.
- ***Moist cool climate***: SOC <u>declines</u> (−19%) under all grazing intensities.
- ***Moist warm climate***: SOC <u>increases</u> (+7.6%) under all grazing intensities.
- ***Dry cool climate***: SOC <u>increases</u> (+16.1%) under low to medium grazing intensities; the effect of high grazing intensity is unknown.

3.19.1.4 Timescale

>10 year
It's unknown how long soil carbon stocks will take to adjust to a change in management, but is likely to take a number of years (Smith, 2014).

3.19.1.5 Spatial Issues

Responses to a change in grazing pressure on carbon stocks are clearly variable in direction and magnitude, and so appropriate targeting of systems likely to benefit from extensive management is key.

3.19.1.6 Displacement

Converting to extensive management has a risk of resulting in the conversion of more land to grazed grassland to maintain agricultural production.

3.19.1.7 Maintenance and Longevity

Soil carbon stocks are sensitive to management pressure and change in response to further changes in management. However, in the absence of changing management, soil carbon is a long term carbon store.

3.19.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation. Reductions in GHGs emitted from grazing livestock could result if displacement is avoided.

3.19.1.9 Climate Factors / Constraints

Reduced grazing in the uplands generally leads to an improved carbon and GHG budget through increased carbon stored in biomass above ground as well as increased recalcitrance of plant litter produced SOC (F. Worrall et al., 2011). Overgrazing results in SOC losses in dry warm and moist cool climates, though it is unclear if reducing the grazing season will be effective in dry cool climates (Abdalla et al., 2018), which may become relevant factors depending on how climate will change in the UK.

3.19.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.19.1.11 Uptake

Grazing is ultimately one of the ways in which farmers make money, so there will be economic barriers to interventions that restrict grazing. In upland areas, while incentives to reduce stocking rates may be readily taken up by some farmers, reducing stocking may run counter to some farmers' ideologies. Furthermore, hefting of sheep on common land may become harder over time with declines in sheep numbers, potentially resulting in a positive feedback and abandonment of some upland areas (Alison et al., 2019). Elsewhere, farmers may be open to incentives for rotational grazing, for instance to enhance plant productivity and by extension increase SOC (Alison et al., 2019).

3.19.1.12 Other Notes

More primary research is needed into determining the optimum grazing pressure on a range of grasslands spanning various configurations of soil type, plant species, grazing management practices and climate. It is not clear that there are consistent definitions of "light", "moderate" or "heavy/extreme" grazing in the literature. Normalised grazing intensity metrics based on regional or global datasets (*sensu* Abdalla et al., 2018) offer one potential avenue to pursue going forward.

3.19.2 **ECPW-032: Use herbal and grass leys**

Duplicated evidence:

Use crops / varieties with differing root structures to improve soil structure and nutrient uptake [**ECPW-265**]; Use grass or encourage natural regeneration where this can be efficiently incorporated into the rotation [**EHAZ-024**]; **Arable02**: Un-vegetated, ploughed fallow (natural regeneration) for one year; **ETPW-202** Plant/ maintain mass flowering crops e.g. legume leys

3.19.2.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**TD***
Global, regional & local climate regulation	Above ground carbon sequestration	**N**
	Below ground carbon sequestration	*****

There is abundant evidence (including from the UK) of the effectiveness of this intervention on alleviating poor soil structure (e.g. compaction), promoting nutrient uptake, and the benefits on SOC sequestration. Key barriers to this intervention include uptake among farmers/land-managers and the timescales typically required from inception to detect tangible benefits on SOC accumulation.

Many of the adverse effects on soils of continuous arable systems stem from a decline in SOM, which in turn degrades soil structure and fertility. SOM is maintained via inputs of root biomass and exudates alongside aboveground plant residues (Blanco-Canqui et al., 2013). This presents a dilemma: crops are selected to maximise photosynthetic gains for the harvestable crop itself, but this will deliver the lowest returns of organic residue to the soil (Gregg et al., 2021).

Logic chains suggest that the adverse effects of agricultural monocrop systems on soil structure can be overcome through the introduction of multiple crops/varieties in a rotation, integration of short-term grass (known as "leys"), and alternating spring and winter crops to manage weeds (Maskell et al., 2019). Much of the evidence for this intervention concerns the use of ley cropping, along with cover cropping (covered in section 3.28.1), appear to be the most extensively studied.

Several primary research studies, reports and meta-analyses indicate that a more diverse root structure yields significant improvements in soil structure and nutrient uptake. Results derived from coastal wetlands reveal plant species richness has been shown to be a significant predictor of root biomass, with secondary effects of promoting carbon storage and reducing erosion (Ford et al., 2016). Ley crops and cover crops promote rooting depth diversity, providing soils with a net gain in SOM – and by extension, SOC. SOC accumulation rates will vary, depending on initial SOC levels in the soil, total inputs of organic matter, rates of decomposition, the duration of the ley, and physico-chemical characteristics of the soil (Johnston et al., 2009). Most of these factors are influenced by the land manager through decisions such as selecting species composition and whether to graze or cut the ley pasture (Gregg et al., 2021 and references therein).

Ley cropping and mixed crop rotations confer other benefits as well. More diverse crop rotations can increase biodiversity both above and below ground; mitigate the impacts of failing/poor performing crops and improve yield stability; disrupt pest and disease cycles; enhance soil quality and fertility; and improve water quality by facilitating enhanced uptake of soil nutrients and reducing nitrogen leaching (Maskell et al., 2019 and references therein). The incorporation of legumes into arable rotations can improve productivity in the absence or reduction of fertiliser inputs, as a result of in situ nitrogen fixation (De Deyn et al., 2011).

The magnitude of effect on sequestration in arable systems that can be expected from this has been widely discussed, but natural regeneration is likely to occur more slowly, and may involve smaller benefits for soil erosion as a result.

3.19.2.2 Co-Benefits and Trade-offs

No assessment.

3.19.2.3 Magnitude

Fornara and Tilman (2004) measured carbon and nitrogen accumulation in agriculturally degraded soils of Montana, USA over 12 years to a depth of 1 m. They found that high-diversity mixtures of perennial grassland plant species stored 500% and 600% more SOC and N than, on average, did monoculture plots of the same species; also the presence of C4 grasses and legumes increased soil C accumulation by 193% and 522%, respectively (Fornara & Tilman, 2008).

Increases in SOC concentrations of 0.36 to 0.59 t C ha^{-1} yr^{-1} were found in a ley dominated rotation compared with a cereal monoculture rotation after 35 years of management (Börjesson et al., 2018). A shorter-term study covering 7 years found more modest results, with intercrop systems associated with -0.184 ± 0.086 t C ha^{-1} yr^{-1} more sequestration than monocrop systems (Cong et al., 2015).

Globally, soils with a cover crop were found on average to have 1.11 t C ha^{-1} more than those with no cover crop (McClelland et al., 2021).

3.19.2.4 Timescale

5-10 years typically – partly because of the length of time some studies ran on average; partly because of the time required to properly establish diverse crops / varieties with differing root structures.

Many of the studies reviewed by McClelland et al. (2021) covered timescales between 5 and 10 years, with some >10 years. The mean timescale of studies reviewed in another meta-analysis was 18 years, though the range was 3-98 years (McDaniel et al., 2014) so the mode may be around 10 years or less and the mean is likely skewed by the most extreme timescale values of the compiled dataset.

Cong et al. (2015) detected modest advantages of intercrop over monocrop systems for storing SOC within 7 years.

3.19.2.5 Spatial Issues

Due to periodic cultivation, SOC accumulation will be comparatively lesser in arable than in improved grassland (Johnston et al., 2009). Other factors that influence the rate of SOC sequestration include nitrogen inputs, soil cultivation practices and soil texture (Johnston et al., 2017). A global meta-analysis of carbon allocation from crops to soils found that the carbon detected in arable soils with a high clay content was almost 150% higher than under sandy soils (Mathew et al., 2020). Higher carbon inputs and storage occur under grazing than mowing management of leys (Gilmullina et al., 2020). Johnston et al. (2017) demonstrated for a sandy soil in south-east England, the SOC increase rate was 9 % higher than the baseline scenario for leys that were grazed, compared with +5 % for stockless leys. Incorporating cover crops appears to be particularly effective, with one global meta-analysis showing that adding one or more crops in rotation to a monoculture increased total SOC by 3.6 %, but the figure for including a cover crop was more than double this at 8.5 % (McDaniel et al., 2014).

3.19.2.6 Displacement

If leys are replacing productive rotations, displacement may occur. However, if the use of leys results in a subsequent increase in productivity, this could be prevented. The overall outcome will involve a complex interplay between the lost rotations of arable productivity, the effect of the ley on subsequent arable yields and any changes in ruminant density or distribution associated with the grazing of the ley. A full review of the potential outcomes of these interactions is beyond the scope of this review.

3.19.2.7 Maintenance and Longevity

The maximum annual rates of C sequestration for soil in ley-arable rotations are estimated at -0.26 to -0.36 t C ha^{-1} yr^{-1} (Johnston et al., 2017) to -1.1 t C ha^{-1} yr^{-1} (Christensen et al., 2009), though these depend on the duration of the grass ley in the arable rotation (Gregg et al., 2021) and the soil type and local climate. Gregg et al. (2021) suggested that about half of the SOC accumulation potential of soils in ley-arable rotations is reached after approximately 6 years (range: 4-11 years). One study has also suggested that integrating leys into all-arable rotations will support sustained or even improved productivity and increases in SOC accumulation over a period of 20-40 years (Knight et al., 2019). While it is possible to maintain a consistent increase in SOC following the introduction of leys, this rate will level off as equilibrium is reached, and because of the impact of periodic cultivation, SOC accumulation will be less in arable than in improved grassland soils (Johnston et al., 2009). Gains in SOC sequestration are also reversible should land use switch back to continuous arable production (Gregg et al., 2021).

3.19.2.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

Rollett and Williams (2020) suggest that increased root mass, could increase plant tolerance to both drought and flooding. They also highlight research by the Salk Institute's Harnessing Plants Initiative which uses genetic and genomic techniques to breed plants so they are optimised to store more carbon naturally[5]. In another study, grass-clover leys were found to regenerate earthworm populations, reduce bulk density, and increase infiltration rates, macropore flow and saturated hydraulic conductivity, thus reducing flood and drought risks to these soils and the wider catchment areas (Berdeni et al., 2021).

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[5] www.salk.edu/science/power-of-plants/

3.19.2.9 Climate Factors / Constraints

For one study comparing a ley-dominated rotation with continuous cereal cropping, it was found that SOC increased in both the topsoil and subsoil in a loam soil, but only in the topsoil of a clay soil (Börjesson et al., 2018). It has been suggested that high bulk- density, anoxic conditions or lack of available phosphorus in the subsoil prevented deeper rooting in the clay soil (Gregg et al., 2021).

3.19.2.10 Benefits and Trade-offs to Farmer/Land-manager

The consensus among the studies reviewed here is that using crops / varieties with differing root structures will lead to increased crop yields between ley rotations, although arable production will be forgone during ley years. Crop rotations can also disrupt pest and disease cycles, potentially reducing the need for pesticides and herbicides (Maskell et al., 2019 and references therein). Longer term improvements in soil fertility brought about by improved soil structure and nutrient uptake should reduce fertiliser costs in the future.

3.19.2.11 Uptake

Introducing new crops requires suitable machinery for sowing and harvest (Maskell et al., 2019). Moreover, knowledge gaps mean there could be lower yields when a new crop is grown for the first time (DEFRA, 2016). In the case of plant breeding, producing a new plant variety can take several years or even decades (Rollett & Williams, 2020).

3.19.2.12 Other Notes

It would be useful to see more primary research on the combined effects of this intervention with minimum-/no-tillage cultivation, cover cropping and controlled traffic farming.

3.19.3 ECPW-022EM: Enhance or manage species-rich grassland habitats;

Duplicated evidence: Enhance and manage flower-rich and species rich grass margins, field corners, and plots [**ETPW-205EM**]; **ETPW-217**: Create areas of scrubby flower-rich grassland; Enhance and manage locally distinctive flower rich/hay meadows using traditional techniques [**EBHE-214EM**]; **EHAZ-010Y** Enhance or manage permanent grasslands.

3.19.3.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**LT***
	Below ground carbon sequestration	**LT***

This action is assumed to apply to permanent grassland only. For the effect of incorporating diverse leys and temporary grassland see section 3.19.2.

Positive effects on SOC have been widely reported to result from biological nitrogen fixation resulting from the incorporation of legumes in grassland, and the inclusion of deep rooted perennials (Alison et al., 2019; Maskell et al. 2019). The majority of carbon inputs to grassland soils come from root structures, and so the inclusion of deeper rooted perennials may lead to an increase in carbon stocks at depth. Increased grassland productivity has also been reported following an increase in sward diversity. Biological nitrogen fixation could also reduce the need for artificial fertiliser application, and therefore provide CO_2-eq abatement.

Dawson & Smith (2007) Reviewed the impacts on land use on soil carbon, and estimated that improving grass species could result in -3.04 10^3 gC ha^{-1} yr^{-1}, or -0.00304 tC ha^{-1} yr^{-1} based on two sources, although and increase in sequestration would not be indefinite and soils would reach saturation.

(Fornara & Tilman, 2008) studied the change in soil carbon content to 60cm depth in a long term (12 years) grassland experiment in Minnesota, USA, as a result of artificially maintained grassland communities. They found that the presence of C4 grasses and legumes increased rates of soil carbon accumulation by 193% and 522% respectively. This study took place in a system which is annually burned, and in abandoned arable land. De Deyn et al. (2011) carried out a 16 year experiment in which grassland biodiversity was manipulation via seeding and manipulation of legume density, at a site in the UK. They found that biodiversity restoration practices and the promotion of the legume *Trifolium pratense* resulted in the sequestration of an additional 317 g C m^{-2} yr^{-1}, or 3.17 tC ha^{-1} yr^{-1}. Over 16 years, this suggests an additional 50.7 tC ha^{-1} was sequestered, which is more than the existing carbon stocks in arable grassland reported by the Countryside Survey 2007 (Emmett et al., 2010). De Deyn et al. (2011) do not report how this rate of sequestration varied over the 16 year period. (Gregg et al., 2021) discuss that this measurement is over 5 times greater than the average carbon sink for European grassland, and that further research is needed to understand how the approach to restoring plant communities and site context impacts the effect on soil properties.

Given our knowledge of carbon stocks in grassland at depth (Bradley 2005 estimate average stocks of 160 tC ha^{-1} for the UK to 1m), it is not likely that such high rates of fixation to persist. Rapid increases in carbon following a change in management can be conflated with legacy effects of grassland soils recovering from historic land use or historic conversion from arable land (Smith, 2014). As a result, the ultimate change in soil carbon stocks that increasing grassland diversity is likely to achieve remains unclear. Maskell et al. (2019) also highlights that the benefits of diversifying grassland species for carbon sequestration could be offset if increased vegetation productivity is translated to a higher stocking density, without improvements in livestock performance.

3.19.3.2 Co-Benefits and Trade-offs

N_2O from legumes in damp conditions
Greater numbers of livestock supported from more productive sward – fab review

3.19.3.3 Magnitude

Estimates of the size of the effect of grassland diversification on carbon sequestration are highly variable and expert opinion suggests that larger estimates are unlikely to be widely observed, or for long periods of time. Dawson & Smith (2007) reviewed the impacts on land use on soil carbon, and estimated that improving grass species could result in -3.04 10^3 gC ha^{-1} yr^{-1}, or -0.00304 tC ha^{-1} yr^{-1} based on two sources, whilst De Deyn et al. (2011) observed a maximum increase in sequestration rates of 317 g C m^{-2} yr^{-1}, or 3.17 tC ha^{-1} yr^{-1} in the most effective treatment.

Conant et al. (2017) estimated that sowing legumes in grassland increased C sequestration rates by a mean rate -0.66 t C $ha^{-1}yr^{-1}$, over 13 studies with a mean duration of 8.3 years.

3.19.3.4 Timescale

Unknown. Evidence from a long term study of land use change at Rothamsted, UK suggest that soil carbon may take over 100 years to saturate following conversion from arable systems to grassland (Johnson 2009).

3.19.3.5 Spatial Issues

No assessment.

3.19.3.6 Displacement

No assessment.

3.19.3.7 Maintenance and Longevity

Soil carbon stocks have a relatively slow response time to changes in management, but are vulnerable to soil erosion. Maintaining vegetation cover and grassland diversity long term will likely be necessary for the full potential of this management to be reached.
Maintaining target diversity and community may also require repeat interventions.

3.19.3.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.19.3.9 Climate Factors / Constraints

No assessment.

3.19.3.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.19.3.11 Uptake

No assessment.

3.19.3.12 Other Notes

None

3.20 Restoration, management and enhancement – Hedgerows

3.20.1 ECCM-025EM: Enhance/ manage hedgerows

Duplicate evidence: enhance/ manage hedgerows around point-source polluters [ECCM-080EM].

3.20.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	N
Global, regional & local climate regulation	Above ground carbon sequestration	L**
	Below ground carbon sequestration	L*

Hedgerows can store significant carbon and require regular management to maintain their density and extent. Maintaining hedgerow density and diversity whilst allowing structures to grow taller and wider where possible will increase total carbon sequestered in hedgerows. Combining required trimming and management of hedgerows with biochar production would reduce carbon lost in this process, and instead convert it to a more secure form of long term storage.

The 2007 Countryside Survey reports 477,000 km of hedgerows in Great Britain and 402,000 km in England and showed a 6.2% reduction in the length of managed hedgerows in Great Britain between 1998 and 2007 (Carey et al., 2008).

Management can have a significant effect of the size of carbon stocks maintained in hedgerows. Axe et al. (2017) conducted one of the largest studies to date on hedgerow carbon stocks in the UK, and showed that hedgerows maintained in taller and wider states contained more carbon in above ground biomass, with a

4.2m wide hedge containing 9.7 tC km^{-1} more than a hedge 2.6m wide, with a mean height of 3.5m. Based on these findings Prosser et al. (2022) reports that increasing hedgerow height from 2.0 m to 2.7 m would represent an increase in size to 70 per cent of currently managed hedgerows across England and Wales, with a potential to sequester an additional 2.0 Mt carbon. When biomass is removed during management, sequestering that carbon in biochar would reduce the loss of sequestered carbon and convert stocks to a more stable form for long-term storage (Gregg et al., 2021) athough there will be practical barriers to implementing this practice at scale. As discussed in Gregg et al. (2021), further research has suggested that hedgerows established for biodiversity, with higher species richness, have higher below ground carbon stocks from 0-1m (175.9 ± 13.2 t C ha^{-1}) than remnant hedgerows (132.7 ± 7.3 t C ha^{-1}).

The effects of these managements on rates of carbon sequestration are not well established, but it is likely that sequestration rates will be highest in younger hedgerows or those responding to recent thinning, and will slow as hedgerows increase in age, based on our understanding of other woody vegetation (Matthews, 2020). The greatest benefit from hedgerow management can be achieved where existing hedgerows are degraded and have lost their density (Axe et al., 2017; Norton et al., 2012).

3.20.1.2 Co-Benefits and Trade-offs

No assessment.

3.20.1.3 Magnitude

Axe et al. (2017) showed a 4.2m wide hedge contained 9.7 tC km^{-1} more than a hedge 2.6m wide, with a mean height of 3.5m. Based on these findings Prosser et al. (2022) reports that increasing hedgerow height from 2.0 m to 2.7 m would represent an increase in size to 70 per cent of currently managed hedgerows across England and Wales, with a potential to sequester an additional 2.0 Mt carbon.
A 30-40% increase in the extent of hedgerows estimated to lead to between a change in emissions of -1 and -4 Mt CO_2eq by 2050, assuming some of the extent is managed for biofuel production, when considered in concert with the effects of agroforestry (Thomson et al., 2020).

3.20.1.4 Timescale

Regular trimming is typically carried out over the time scale of 1-3 years, with more detailed maintenance after a period of roughly 40 years (Gregg et al., 2021).

3.20.1.5 Spatial Issues

Likely to have successful broad implementation, with consideration given to appropriate soil type and species.

3.20.1.6 Displacement

No assessment.

3.20.1.7 Maintenance and Longevity

Management would need to be ongoing to preserve benefits.

3.20.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.20.1.9 Climate Factors / Constraints

No assessment.

3.20.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.20.1.11 Uptake

No assessment.

3.20.1.12 Other Notes

None

3.21 Restoration, management and enhancement – Mountain, moor and heathland

3.21.1 EBHE-216: Enhance or manage moorland (including common land), e.g. through appropriate traditional grazing techniques

Duplicated evidence: Only burn in accordance with the heather and grass burning code. [**ETPW-144**] Enhance or manage heathland (including heathland mosaics) [**ECPW-176EM**]; Where burning takes place, ensure small burns on a long rotation to create a varied age structure in dwarf shrub, including retaining mature and degenerate phases [**ETPW-143**]; Off-winter livestock or reduce winter grazing on upland and mountain heath [**ETPW-142**]

3.21.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**TD****
	Below ground carbon sequestration	**TD***

There is good evidence that overall rates of carbon sequestration are relatively high in moor and heath, and that soil carbon stocks can also be moderate, but the impacts of traditional management on heath and moor carbon are contentious, and contradictory evidence exists in the literature. The largest positive impact on above ground biomass could likely be achieved through reduced burning frequency. Other actions may be associated with a smaller impact.

Excessive nutrient deposition, over grazing and inappropriate burning regimes can result in the loss of heathland ecosystems and degradation to grassland, which is associated with lower rates of carbon sequestration (Gregg et al., 2021). The development of bare ground can also result in soil carbon loss via erosion and absence of organic inputs from plants. Among management intensities that do not result in habitat degradation, there is no consensus what level of intensity is best suited for the promotion of carbon stocks and sequestration.

<u>Grazing</u>

Livestock counts for sheep in UK moorland are thought to have increased by approximately 30% from 1970 to 1990, driven by EU Common Agricultural Policy subsidies based on headage (Alison et al., 2019; Worrall et al., 2011). Alison et al. (2019) reviewed the evidence for an effect of grazing pressure on soil carbon in the uplands in Wales and found contradictory evidence, with some studies indicating low grazing pressure typical of extensive farming would increase soil carbon stocks (Smith, 2014), whilst others suggested the intensification of upland farming may have some positive effects on soil organic carbon, for example by increasing rates of root exudation in response to grazing, and others found reducing sheep grazing pressure had no effect on soil chemistry and vegetation biomass (Robert H. Marrs et al., 2018). However, for an increase in grazing pressure to be carbon neutral, and increase in headage would need sequestration rates to increase by a sufficient amount to offset the emissions from the livestock themselves, which is not accounted for in the evidence reviewed by Alison et al. (2019). Modelling work by Sozanska-Stanton et al. (2016) shows that grazing dwarf shrub heathlands results in a small increase in emissions per animal, of +0.1

tCO_2 eq ha^{-1} yr^{-1}, but their analysis uses parameters derived from farm livestock, including emissions due to manure disposal and feedstock provision, as pointed out by (Gregg et al., 2021).

Burning

The burning of heathland is often used as a tool to maintain a specific vegetation community or promote regrowth which leads to a preferable food source for grouse rearing (Gregg et al., 2021). Burning in heathland is legally regulated, an addition to good practice being set out in the Heather and Grass Burning Code (Heather and grass burning: rules and applying for a licence - GOV.UK (www.gov.uk)), but remains highly controversial (Davies et al., 2016). Legislation covers the timing of burning, the areas that can be burned and specific areas that should be excluded from burning. Assuming fires do not damage the soil layer, they will result in the net emission of carbon due to combustion and the loss of photosynthesis in the short term, whilst (small) emissions of N_2O and CH_4 continue. In the long term, those vegetation stocks will be rebuilt, but if they are regularly burnt (recommended burning frequency is 10 years) then above ground sequestration cannot be considered a net gain.

An analysis of the effects of burning *Calluna* heathland following existing guidance and Tier 1 emissions factors from the IPCC suggested that recommended burning resulted in the loss of 6.911 tCO2eq ha^{-1} yr^{-1} (Carey et al., 2015). A decrease in management frequency would result in larger, overall carbon stocks at a landscape scale (Gregg et al., 2021), but sequestration rates may also reduce. Any increase in burning frequency leading to reduced carbon stocks at the landscape scale will accrue a carbon debt relative to historic conditions, as with an increase in management intensity in woodland (Matthews, 2020). Reducing the frequency or intensity of managed burns has been suggested to increase the risk of wildfire, although the interactions between managed burns and wildfires is complex, particularly over peat where stakes are particularly high (Davies et al., 2016) and burning for wildfire risk management remains contentious (Gregg et al., 2021). Despite this logic chain, a previous report on the impacts of controlled burning in the Welsh uplands reported that effects in the literature were unclear, with studies disagreeing in both the magnitude and direction of effect (Alison et al., 2019). The majority of studies compare the effects of burning with no burning on peatland, with less evidence for the effects of reduced frequency burning, and response metrics are variable not directly comparable between studies.

Overall, there is some evidence and a consistent logic chain that reducing burning frequency will result in net C sequestration and storage at a landscape scale, although confidence remains low and further research into the responses of different vegetation communities is desirable. In addition, Grau-Andrés et al. (2018) and Gregg et al. (2021) recommend controlled fires are restricted to times when moss and litter fuel moisture content is over 150 % and soil fuel moisture content is over 200-300%, to ensure low burn temperatures, to minimise the risk of additional carbon loss from soil.

Draining

Where heath or moorland occurs on degraded peat, the best management course for net carbon sequestration and storage is to restore the peatland, and raise the water table in particular (Evans et al., 2017).

3.21.1.2 Co-Benefits and Trade-offs

No assessment.

3.21.1.3 Magnitude

Reduce grazing intensity:

Modelling work by Sozanska-Stanton et al. (2016) shows that grazing dwarf shrub heathlands results in a small increase in emissions per animal, of +0.1 tCO_2 eq ha^{-1} yr^{-1}, but their analysis uses parameters derived from farm livestock, including emissions due to manure disposal and feedstock provision, as pointed out by

(Gregg et al., 2021). Other research suggests a negative or neutral effect of reducing grazing intensity on mountain. Moor and heath.

Reduced burning frequency:
Sozanska-Stanton et al. (2016) estimate that stopping the prescribed burning of heathland and moor grass would reduce GHG emissions by 7 t CO_2eq ha^{-1} y^{-1}, using IPCC methods. Impacts due to wildfire risk are less certain. Reduced burn frequency may have impacts on the vegetation community and biodiversity long term.

Restore water table on peat:
Evans et al. (2017) estimated of an emissions factor for rewetted bog of 0.81 tCO_2e $ha^{-1}yr^{-1}$. This is compared to estimated emissions factors of +19.0 tCO_2e $ha^{-1}yr^{-1}$ for extensive grassland, and between +4.85 tCO_2e $ha^{-1}yr^{-1}$ and +2.08 tCO_2e $ha^{-1}yr^{-1}$ for modified fen and bog.

3.21.1.4 Timescale

Changes in management, particularly burning, can have an immediate effect on carbon sequestration rates and stocks. However, stabilisation in carbon stocks following a change in management may take many years to occur, particularly for peatland restoration.

3.21.1.5 Spatial Issues

No assessment.

3.21.1.6 Displacement

If livestock headage is reduced, there may be some risk of displacement to other habitats or internationally, due to continued demand.

3.21.1.7 Maintenance and Longevity

Any changes in management would need to be maintained long-term for associated changes in carbon to persist.

3.21.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.21.1.9 Climate Factors / Constraints

No assessment.

3.21.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.21.1.11 Uptake

Changes to heathland management that are likely to increase carbon sequestration and storage may run counter to the needs of management for biodiversity, and may impact the revenue generation from upland sites, particularly those associated with livestock rearing.

3.21.1.12 Other Notes

None

3.22 Restoration, management and enhancement – Riparian areas

3.22.1 ECPW-291C: Create riparian habitats

Duplicated evidence:
ECPW-042: Create/ enhance/ manage riparian buffer strips; **ECPW-291EM**: Enhance or manage riparian habitats; **ETPW-038**: Create/ manage/ enhance buffer strips; **ECPW-157C**: Create buffer strips (including trees) around boreholes; **ECPW-157EM**: enhance/ manage buffer strips (including trees) around boreholes

Co-Benefits and Trade-offs only

3.22.1.2 Co-Benefits and Trade-offs

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**T****
	Below ground carbon sequestration	**T***

Creating riparian habitat can result in carbon sequestration, where land is taken out of arable production, and particularly if woody vegetation in planted. A more detailed review of the potential carbon benefits from creating a variety of habitats, including diverse grassland, scrub and woody features can be found in section 3.5. There is less evidence for the effect on soil carbon stocks of taking permanent grassland out of production to form a buffer strip. However, livestock with unrestricted access to water courses can cause significant poaching and sediment loss to erosion, which may be prevented by the creation of a fenced buffer strip (see section 3.27.3).

3.23 Restoration, management and enhancement – Scrub

3.23.1 EBHE-203EM: Enhance manage targeted scrub

Duplicated Evidence: **ETPW-112** Manage scrub to maintain, restore and enhance grassland condition and associated species populations, recognising its inherent value in providing shelter/structure/food and nesting resource

3.23.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**LT***
	Below ground carbon sequestration	**LT***

Evidence for the impacts of scrub management on carbon sequestration are lacking. Based on evidence from other systems containing woody vegetation, management that are likely to affect carbon stocks in scrubland include the frequency and intensity of any grazing and cutting and resultant changes to the biomass of scrub, which can result in further carbon sequestration above and below ground (Gregg et al., 2021). The management of scrub to facilitate further secondary succession to woodland is likely to result in the largest potential for carbon sequestration and storage. Natural regeneration from abandoned agricultural land to woodland can take many decades in the absence of assistance (Gregg et al., 2021; Nadal-Romero et al., 2016). Allowing an increase in scrub area and density in place of grassland will also increase carbon stocks in above and below ground biomass (Gregg et al., 2021). The extent of scrubland is usually managed to either maintain its extent, when scrub is of conservation value, or constrain its extent when scrub is considered undesirable. Responses are likely to vary significant across types of scrub.

3.23.1.2 Co-Benefits and Trade-offs

No assessment.

3.23.1.3 Magnitude

Allowing the succession of scrub to birch woodland has been estimated to sequester approximately -2 t C ha^{-1} yr^{-1} (reported in Gregg et al., 2021; original by Uri et al., 2012).

3.23.1.4 Timescale

Dependent on management intervention – succession may take > 10 years, however some changes in carbon stocks will the apparent faster.

3.23.1.5 Spatial Issues

No assessment.

3.23.1.6 Displacement

No assessment.

3.23.1.7 Maintenance and Longevity

Carbon stocks could be lost by an intensification of management.

3.23.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.23.1.9 Climate Factors / Constraints

No assessment.

3.23.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.23.1.11 Uptake

No assessment.

3.23.1.12 Other Notes

None

3.24 Restoration, management and enhancement – Woodland

* Duplicated evidence base: managements for carbon sequestration are the same across these Manage large-scale woodland in priority catchments [**ECCA-018EM**]; Manage or enhance targeted woodland [**ECPW-044Y**]; Enhance/ manage Ghyll woodland [**EBHE-140EM**]; Enhance or manage floodplain woodland [**ECPW-071Y**]; Use woodland management (UKFS) for target priority woodland species [**ETPW-266**];

3.24.1 EBHE-314: Create a woodland management plan

Duplicated evidence: Create and use a contingency plan that integrates tree / plant pests and diseases with other threats (flooding, drought etc.) [**ETPW-120**]

3.24.1.1 Causality

Global, regional & local climate regulation	Above ground carbon sequestration	T
	Below ground carbon sequestration	T

Although the effects of using a woodland management plan on carbon sequestration have not been evaluated in the literature, there is a consensus opinion that detailed consideration of the objectives behind tree planting, appropriate methodology, location, and the development of an adaptive management plan are strongly associated with successful tree planting and initiatives (Brancalion & Holl, 2020). The consideration of these issues, and the requirement for monitoring and reporting are also park of the UKFS. The value of increasing woodland carbon stocks for climate change mitigation is contingent on the preservation of those stock as sequestered carbon long term (Matthews, 2020). Strategies, such as using a woodland management plan, increase the likelihood of long term woodland persistence are therefore likely to be beneficial for carbon sequestration. The consideration of long-term woodland management strategy and putting in place an adaptive management plan are desirable components of a woodland management plan (Brancalion & Holl, 2020), as management has a significant effect on the resilience of woodland to future pressures and it's carbon sequestration and storage potential (Matthews, 2020). However, in the absence of financial support for long term management, planning may not yield rewards.

Management plans should be a central part of any woodland creation plan to ensure desired outcomes are achieved (see section on 'Habitat Creation – Woodland'). However, for management plans to be effective they require the continued availability of the necessary financial support for long term and adaptive planning.

3.24.1.2 Co-Benefits and Trade-offs

[TOCB Report-3-5D Systems **EBHE-314**] Woodland management, in particular, can benefit from the use of a management plan, to coordinate the extraction of timber from felling and coppicing, maintenance of rides, fencing to control deer, control of Rhododendron, etc., with a view to maximising habitat value at the level of a whole site, or landscape.

[TOCB Report-3-5C Semi-natural **EBHE-314**] Causality Scored as (green T) as for all plans, also scored as for woodland creation **EBHE-104**. To mitigate disbenefits from woodland expansion on biodiversity, site-based evaluations are necessary, careful forest design planning and tailored management of new woodland sites. Expert value judgements may be required to establish which elements of biodiversity and ecosystem services are prioritised at both local and national scale. However, these local judgements must sit within in a strategic landscape, regional and national framework to ensure all habitats are conserved (Beauchamp et al. 2020). No evidence was found regarding biodiversity benefits resulting from the creation of a woodland management plan alone.

3.24.1.3 Magnitude

The impact of using a woodland management plan on carbon sequestration and storage has not been quantified.

3.24.1.4 Timescale

The impact of a successful woodland management plan would be felt at a range of timescales, by (for example) increasing the likelihood of successful forest establishment, minimising negative consequences from land use change, and supporting the long term condition of the woodland.

3.24.1.5 Spatial Issues

No assessment.

3.24.1.6 Displacement

No assessment.

3.24.1.7 Maintenance and Longevity

The long term support of management and monitoring for created woodland is key to the maintenance of carbon stocks.

3.24.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.24.1.9 Climate Factors / Constraints

No assessment.

3.24.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.24.1.11 Uptake

Guidance on creating a woodland management plan is well established[6]. The availability of financial support to enable long term management plans to be put in place and followed is also important to translate a plan into a successful outcome.

3.24.1.12 Other Notes

n/a

3.24.2 **ETPW-125: Coppice and thin trees**

Duplicated evidence: **ETPW-019**: Use coppice for bank reinforcement.

3.24.2.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**TD***
Global, regional & local climate regulation	Above ground carbon sequestration	**LTD***
	Below ground carbon sequestration	*

Thinning

The impacts of thinning are complex, with the effects on carbon sequestration dependent on the end use of harvested biomass, yield class and ages. Thinning can lead to a small reduction in the mitigation potential of a woodland for carbon emissions, but might be necessary for the long term health of overstocked stands and wider management objectives. It is also changes the distribution carbon across possible stores, and is associated with the loss of carbon in soils, whilst more carbon is stored in wood products or as a result of reduced fossil fuel emissions due to product substitution. For existing stands that are thinned under a consistent management regime, reducing management intensity will likely result in an increase in overall carbon sequestration, whereas increasing thinning intensity or introducing thinning to unthinned stands will likely result in a net emission of carbon. Stands that are newly created and managed for thinning can be significant carbon sinks (Matthews 2020; Matthews et al. in prep.).

It is well established that the thinning of a stand results in lower overall sequestration rates compared to those in un-thinned stands, despite stimulating higher growth rates in the remaining trees, due to reduced competition (Matthews et al., in prep. / 2020). Stocks of carbon in woodland are reduced significantly in the

[6] www.gov.uk/guidance/create-a-woodland-management-plan

short term, and assuming regular thinning, would be kept at a lower average level than un-thinned stands long-term. Matthews (2020) show the long-term mean carbon stock in thinned woodland was reduced to 40 tC ha^{-1} ,compared to 60 tC ha^{-1} in an un-thinned control (Matthews, 2020). Thinning has also been advocated for on the grounds that it may increase stand resilience to drought, due to reduced below ground competition (D'Amato et al., 2013).

The woody biomass produced by thinned and un-thinned stands can be roughly equivalent, following good management, as shown by data reported in Matthews et al. (in prep.). As trees in thinned stands tend to have larger basal area, they may be more suited for the production of long-lived wood products than un-thinned stands for commercial production (Matthews et al., in prep.). The negative consequences of thinning for soil carbon stocks however are potentially substantial, and can cause the continued net loss of soil carbon stocks throughout the full vigour phase of growth. In a long term study of the effect of thinning on woodland carbon stocks, Matthews et al., (in prep.) report that thinned stands had a persistent net loss of soil carbon at a rate on 1.85 tC ha^{-1} yr^{-1} between the ages of 6 and 35, which resulted in the net loss of soil carbon since beginning site preparation of 8.25 tC ha^{-1} by age 35. For comparison, the un-thinned site had achieved the net sequestration soil carbon by -4.12 tC ha^{-1} by age 35. Henneron et al. (2018) report that higher thinning intensities in *Quercus petraea* significantly reduced the rate of above ground litter inputs in stands ages 0-95, but not for stands ages 95-165. Stands also responded to thinning by altering the nutrient content of leaf litter, which resulted in slower rates of decomposition where the rate of litter fall was lower, leading to a small decrease in topsoil carbon stocks with thinning intensity over all.

The motivation for thinning is typically economic or for conservation purposes, and often occurs at the marginal intensity (the maximum volume that can be removed periodically without causing a loss of cumulative total volume over the rotation) (Matthews et al., in prep.). Where avoiding thinning is not an option, forests that are thinned can still contribute signficant carbon sequestration overall.

Coppicing

Coppicing is a management technique that involves removing a large proportion of biomass from an individual tree to form a low stump, and subsequently allowing it to regrow. Species that are regularly coppiced in the UK include willow, poplar, sweet chestnut and hazel. Coppiced material is typically used for wood products, or to maintain traditional conditions for biodiversity, but short rotation coppice is also promoted for use in bioenergy. For a review of the use of bioenergy crops see section 3.4. In addition to the coppicing of woodland, coppiced vegetation may also fall under the definition of hedgerows and scrub (Gregg et al., 2021). As is the case for thinning, coppicing will reduce the long-term average carbon sequestration potential of a woodland compared to an un-coppiced system, with the magnitude dependent on the intensity and frequency of the coppicing, although notably less is known about the overall trade-off for GHG mitigation from coppicing than for thinning. Coppicing will also change the distribution carbon across possible stores, and is likely associated with loss of carbon in soils (although evidence is scarce), and more carbon sequestered in wood products or as a result of product substitution. Reducing management intensity in woodlands that have a history of coppice will likely increase long-term carbon stocks above and below ground, whereas increasing coppicing intensity or introducing coppicing to stands will result in an emission of carbon, or carbon debt (depending on the fate of wood products). Stands that are newly created and managed for coppicing can be significant carbon sinks, particularly if soil carbon loses are minimised (Matthews 2020; Matthews et al, in prep).

There is a lack of studies in which coppiced stands are compared to an un-coppiced control, in addition to a lack of data about biomass retained in coppiced root systems and stools. The sequestration potential of coppice systems long term is dependent upon the survival of coppice stools, which is reportedly variable, but in most survive to restore levels of canopy cover prior to intervention, when associated natural regeneration between coppices is accounted for (Buckley, 2020). Matthews (2001) modelled the carbon allocation and energy budgets for a coppice system used for wood fuel. They reported that between -5 and -12 tC ha^{-1} were sequestered of a 25 year period in un-used biomass. System biomass accumulated monotonically with stool age, despite large fluctuations in standing biomass with coppicing cycles, although the technical saturation point is significantly lower than would be expected in the absence of management.

These estimates of net sequestration do not include any assessments of the impact on emissions due to the lifecycle of products. The analysis also had high sensitivity to assumptions about farm practices and processing and energy costs, and the assumption that coppice production is sustainable long term, without a drop in yield. Pietrzykowski et al. (2021) report the above and below ground woody biomass sequestration in vegetation biomass was reported for a willow coppice in Poland over 12 years. Total carbon sequestered (above-ground biomass, coarse roots and fine roots) was estimated at -13.5 Mg C ha^{-1} yr^{-1}, but changes in soil carbon stocks and wider life cycle emissions are not included.

3.24.2.2 Co-Benefits and Trade-offs

[**ETPW-019**] There is good evidence the growth of new trees for coppicing and subsequent regeneration will result in the sequestration of above and below ground carbon, although changes in below ground carbon stocks with tree establishment on grassland are less clear, and there may be a short term loss of soil carbon associated with establishment (Matthews, 2020; Prosser et al. 2022). For a full review of the effect of coppicing existing trees (compared to not coppicing) and its effect on the potential for carbon sequestration in woodland see section 3.24.2.

Food and fibre production	Area under production or yield and outside of ELM	N
Global, regional & local climate regulation	Above ground carbon sequestration	**
	Below ground carbon sequestration	LTD**

[TOCB from Report-3-5B Grassland **ETPW-019**] Positive benefits for biodiversity from introduction of woody habitat, especially if native species are used.

[TOCB from Report-3-5D Systems **ETPW-019**] Coppicing can have clear benefits for biodiversity in a woodland context, but it is not clear what the alternative is in this context – if it is to have no woody vegetation, benefits and costs are similar to hedge creation (**ECCM-025C**). If the alternative is unmanaged trees, some species will respond more positively to shorter, denser trees, but with benefits only when the trees are more mature, whereas others will prefer more natural structures.

3.24.2.3 Magnitude

Thinning

Maximum and minimum rates of woodland carbon sequestration (tC ha^{-1} yr^{-1}) were reduced by regular thinning for all yield classes estimated by Matthews et al., in prep., including stock changes in living biomass, deadwood, litter and soil. Results are reproduced from the report for a time horizon of 0-35 years after creation and 36 to 52 years after creation in Table 3. The sign of results has been changed from Matthews et al. (in prep.) to be consistent with the rest of this report where negative values indicate net sequestration and positive values are net emission. Lower yield classes were most at risk of thinning causing net carbon emissions due to soil carbon loss. Sequestration rates in older, high yield conifers was highly variable and showed small or negative rates of sequestration due to clearfelling and variable subsequent recovery during this time period. The carbon emissions mitigation potential (including the net impacts of emissions from management and transport, wood products acting as sinks, substitution for fossil fuels and other materials) of woodlands that are thinned vs unthinned are reproduced for the same yield classes and time horizons in Table 4, where once again negative numbers represent sequestration and positive numbers represent emission, for consistency. Note that the incorporation of these fluxes results in stands managed with clearfelling contributing overall to climate change mitigation (through net GHG emissions savings).

Table 3 Indicative ranges for mean rates of woodland carbon sequestration, taken from Matthews et al. (in prep.) for the periods between 0 and 35 years and between 36 and 52 years from time of woodland creation, with and without thinning.

Yield classes (m^3 ha^{-1} yr^{-1})	Woodland carbon sequestration from 0-35 years tC ha^{-1} yr^{-1}			
	Minimum	Maximum	Minimum	Maximum
	Management with regular thinning		Management without thinning	
2-4	+0.57	-0.66	+0.57	-1.2
6	-0.03	-1.3	-0.19	-2.3
8-14	-0.47	-3.3	-0.82	-5.7
16-24	-2.5	-5.2	-3.9	-7.6
	Woodland carbon sequestration from 36-52 years tC ha^{-1} yr^{-1}			
	Minimum	Maximum	Minimum	Maximum
	Management with regular thinning		Management with regular thinning	
2-4	0.32	-1.7	-0.11	-3.1
6	-2.0	-3.4	-3.4	-5.6
8-14	-0.87	-5.1	-1.56	-8.8
16-24	+4.7	-0.42	+7.6	-0.82

Table 4 Indicative ranges for mean rates of GHG emissions mitigation, taken from Matthews et al. (in prep.), for the periods between 0 and 35 years and between 36 and 52 years from time of woodland creation, with and without thinning.

Yield classes (m^3 ha^{-1} yr^{-1})	Total emissions mitigation from 0-35 years tC ha^{-1} yr^{-1}			
	Minimum	Maximum	Minimum	Maximum
	Management with regular thinning		Management without thinning	
2-4	+0.60	-0.99	+0.60	-1.2
6	-0.13	-2.0	-0.16	-2.3
8-14	-0.69	-5.1	-0.80	-5.7
16-24	-3.2	-6.7	-3.8	-7.6
	Woodland carbon sequestration from 36-52 years tC ha^{-1} yr^{-1}			
	Minimum	Minimum	Minimum	Minimum
	Management with regular thinning		Management with regular thinning	
2-4	+0.07	-2.4	-0.11	-3.1
6	-2.6	-4.0	-3.4	-4.9
8-14	-3.6	-9.0	-4.9	-12.2
16-24	-4.3	-9.3	-5.4	-12.1

3.24.2.4 Timescale

See Table 3 and Table 4.
Coppicing frequency can vary from approximately 3-20 years between species.

3.24.2.5 Spatial Issues

Targeting is critical
Exposure to high winds is a significant limitation to thinning practices (Matthews et al., in prep.). The targeting of appropriate species growing under appropriate conditions has a large effect on net carbon sequestration.

3.24.2.6 Displacement

No assessment.

3.24.2.7 Maintenance and Longevity

No assessment.

3.24.2.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.24.2.9 Climate Factors / Constraints

Exposure to high winds is a significant limitation to thinning practices (Matthews et al., in prep.). The targeting of appropriate species growing under appropriate conditions has a large effect on net carbon sequestration.

3.24.2.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.24.2.11 Uptake

It is argued by Matthews et al., in prep. that other requirements of forest management may provide overriding insentives to thin stands even where negative impacts of carbon sequestration could be demonstrated.

Guidance on good thinning practice is available at The Thinning Control Field Guide (Forestry Commission, 2015)

3.24.2.12 Other Notes

The impact of product substitution on fossil fuels use are difficult to anticipate due to their dependence on consumer demand and wider market forces.

3.24.3 (ECCA-028) [ECCM-054]: Diversify woodland / forest / plantation stand structure and species, including by the use of continuous cover systems of management

Duplicate evidence base: Restore / manage ancient woodland with native broadleaf species [**EBHE-198**]; Planted Ancient Woodland (PAWS) restoration [**EBHE-196**] *; Encourage diversification of the stand and continuity of canopy cover through natural regeneration of native species in semi-natural woodland [**ECCA-027**]; Restock trees for resilience [**ETPW-123**].

3.24.3.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T****
Global, regional & local climate regulation	Above ground carbon sequestration	**LT***
	Below ground carbon sequestration	**LT***

Planted ancient woodland sites (PAWS) restoration involves the restoration of sites where ancient semi-natural vegetation has been replaced with a plantation to a system that resembles native woodland, and potentially taking the woodland out of production. One of the main methods of restoration currently receiving attention involves the use of continuous cover approaches, which can be thought of as an extension of the thinning cycle, in place of clearfelling, paired with assisted regeneration of the desired stand community. The principle benefits of regeneration via a continuous cover system over clear felling, are proposed to include reduced disturbance for biodiversity, reduced disturbance for soil carbon and loss through erosion, maintenance of microclimate buffering and greater control over conditions to facilitate establishment (Matthews et al., in prep.) + (Thompson et al., 2003).

There is a general lack of evidence for the effect of PAWS restoration on rates of carbon sequestration, although studies on the effectiveness of PAWS restoration for biodiversity are currently underway[7]. Based on evidence for woodland systems more generally, and guidance published by Forest Research, the overall effect is likely to vary with the suitability of the site for growth by different species and the existing management and species composition of woodland to be restored (Matthews et al., in prep.; Thompson et al., 2003). As discussed in section 3.12.3, the long term potential for carbon sequestration in native broadleaf or coniferous plantations (with typical yield) are similar, when assessed as tC ha^{-1}, but in practice different species combinations are best suited to different environments across the UK, to maximise carbon sequestration. The success of PAWS restoration, where the desired community and functionality typical of an ancient woodland is achieved, will also be dependent the prior type of woodland and management history, which can result in changes to soil pH, nutrient content, ground flora, presence of invasive species and species represented in the seed bank (Thompson et al., 2003).

There is some evidence that adopting a more diverse forest community can enhance productivity and provide resilience to disturbance. For a review of the effects of stand diversity of resilience, see section 3.4.2.

It is worth highlighting that PAWS restoration may be associated with a change in long term management regime, as a result of a transition from a production woodland to a conservation woodland. This is likely associated with an increase in woodland carbon stocks but a decrease in mean sequestration rate as discussed in the section on 'Habitat Creation – Woodland'. If this is the case there is likely to be displacement in land used for fibre production to other systems.

3.24.3.2 Co-Benefits and Trade-offs

No assessment.

3.24.3.3 Magnitude

See section on 'Habitat Creation – Woodland' for a comparison of the rates of carbon stocks and sequestration rates across native, mixed broad leaf woodland and coniferous plantation.

A continuous cover system has the potential to mitigate the loss of over 10 tC ha^{-1} in soil carbon in the 3 years after clearfelling (Matthews et al., in prep.).

The net carbon balance of a woodland managed for CCF between years 53 and 86 years since establishment was estimated as -188.2 tC-eq ha^{-1} or 5.7 tC-eq ha^{-1} y^{1} (Matthews et al., in prep.). Modelling the carbon balance for the same system with clearfelling and subsequent regeneration (with the same species are the original stand, to achieve the same age since establishment) resulted in an estimated net sequestration of -123.7 tC-eq ha^{-1} or 3.7 tC-eq ha^{-1} y^{1} (Matthews et al., in prep.). These assessments include emissions due to processing, transportation and product use. The difference in estimates for the Continuous Cover Forestry (CCF) woodland and the woodland managed with clearfelling suggests potential net carbon sequestration through transformation of woodlands to CCF of 2 tC-eq ha^{-1} y^{1} (7.3 tCO_2-eq ha^{-1} y^{1}) over a 33 year period. However, the long-term effects of a CCF system on above ground carbon stocks are unclear (Stokes & Kerr, 2009).

[7] www.forestresearch.gov.uk/research/ecology-of-upland-native-woodlands/restoration-of-upland-planted-ancient-woodland-sites-paws/

3.24.3.4 Timescale

>10 years
The time required for full functional restoration of ancient woodland is unknown, it likely to require many decades. Species regeneration may occur faster, with some progress expected to be detectable by monitoring on frequencies of 2-10 years (Thompson et al., 2003).

Any changes in mean above ground carbon stocks are likely to occur over the course of multiple decades, whilst soil carbon stocks may take 100.

3.24.3.5 Spatial Issues

Targeting of areas most likely to have successful restoration and establishing site-appropriate species will be important.

3.24.3.6 Displacement

If restoration occurs in highly productive plantations there may be displacement of wood production nationally or internationally. Alternatively, product substitution could occur, with an associated impact on net emissions.

3.24.3.7 Maintenance and Longevity

The success of restoration is likely to vary across sites and ongoing management may be needed to achieve the desires stand composition and ecosystem functionality. Restoration via continuous cover forestry is likely to be a relatively long term process.

3.24.3.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.24.3.9 Climate Factors / Constraints

The resilience of native forests to climate change is an area of concern, resulting in proposals to select stock with resistance to warmer temperatures and drought stress.

3.24.3.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.24.3.11 Uptake

No assessment.

3.24.3.12 Other Notes

None

3.24.4 ECCA-028: Transform (native and exotic) plantation woodland to continuous cover system of management

3.24.4.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**LTD***
	Below ground carbon sequestration	*****

Continuous cover forestry can be thought of as the indefinite continuation of the thinning cycle, in place of clearfelling, potentially paired with establishment of a more diverse stand community, in both species composition and structure, and may involve the establishment of multiple canopy strata (Fritsche et al., 2020; Stokes & Kerr, 2009). This practice is relatively new in Britain. The principle benefits of adopting a continuous cover system over clear felling, during the transition between woodland communities are proposed to include reduced disturbance for biodiversity, reduced disturbance for soil carbon and loss through erosion and maintenance of microclimate buffering and greater control over conditions to facilitate establishment (Matthews et al., in prep.) and Thompson et al., (2003). The subsequent development of uneven aged stands is also thought to increase resilience to storms and high winds (Matthews, 2020; Stokes & Kerr, 2009). The degradation and loss of soil carbon during clearfelling can be significant, particularly where soils are rich in organic carbon. Matthews et al., in prep. discuss the findings of (Xenakis et al., 2021), where the clearfelling of sitka spruce on peaty gley resulted in the immediate loss of 7.05 tC ha^{-1} yr^{-1}, which negates the mean uptake over the previous four years. In the second year after clearfelling, soils lost 3.06 tC ha^{-1} yr^{-1}, and 0.97 tC ha^{-1} yr^{-1} in the third year. These measurements do not distinguish between different sources and sinks of CO_2. The transition from clearfelling to CCF will also affect the rates of production in forestry systems in both the short and long term. In the short term, thinning rates are typically elevated (particularly if new species are being introduced), whereas in the long term, rates of productivity have been frequently debated. Productivity in both even ages and CCF stands are highly variable, making comparison difficult. There is some suggestion that CCF stands may fix carbon dioxide more efficiently than even aged stands per unit area, although a scientific consensus is lacking (Stokes & Kerr, 2009). Changes in harvesting rate are likely to be site specific, and to be strongly dependent in any change in yield class.

3.24.4.2 Co-Benefits and Trade-offs

No assessment.

3.24.4.3 Magnitude

A continuous cover system has the potential to mitigate the loss of over 10 tC ha^{-1} in soil carbon in the 3 years after clearfelling (Matthews et al., in prep.).

The net carbon balance of a woodland managed for CCF between years 53 and 86 since establishment was estimated as -188.2 tC-eq ha^{-1} or -5.7 tC-eq ha^{-1} y^{-1} (Matthews et al., in prep.). Modelling the carbon balance for the same system with clearfelling and subsequent regeneration (with the same species are the original stand, to achieve the same age since establishment) resulted in an estimated net sequestration of -123.7 tC-eq ha^{-1} or 3.7 tC-eq ha^{-1} y^{-1} (Matthews et al., in prep.). These assessments include emissions due to processing, transportation and product use. The difference in estimates for the CCF woodland and the woodland managed with clearfelling suggests potential net carbon sequestration through transformation of woodlands to CCF of 2 tC-eq ha^{-1} y^{-1} (7.3 tCO_2-eq ha^{-1} y^{-1}) over a 33 year period. However, the long-term effects of a CCF system on above ground carbon stocks are unclear (Stokes & Kerr, 2009).

3.24.4.4 Timescale

>10 years
Transformation of forestry systems to CCF is likely to require many years to complete (Stokes & Kerr, 2009).

3.24.4.5 Spatial Issues

Whilst the principle of CCF is broadly applicable, practical implementation will require the targeted consideration of local conditions and the likely success of establishing a CCF system whilst maintaining productivity, if that is the objective.

3.24.4.6 Displacement

The risk of production displacement will be dependent on whether the transition to a CCF system can maintain similar levels of productivity long term.

3.24.4.7 Maintenance and Longevity

No assessment.

3.24.4.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.
The conversion to a CCF system is thought to confer several benefits in terms of stand resilience to climate change and wind damage (Stokes & Kerr, 2009).

3.24.4.9 Climate Factors / Constraints

Not assessed

3.24.4.10 Benefits and Trade-offs to Farmer/Land-manager

There is a potential for a diversification in wood products, and greater resilience in stands to climate change and therefore long term productivity. However, changes in overall productivity are uncertain.

3.24.4.11 Uptake

The initial transition to CCF may require specialist support, top put in place a site appropriate plan. Any change in wood products that can be produced from stands may also provide a challenge to uptake, if new markets must be identified.

3.24.4.12 Other Notes

None

3.24.5 ECCM-053: Manage deadwood (where appropriate, remove diseased deadwood, leave healthy deadwood to contribute to carbon storage)

3.24.5.1 Causality

Global, regional & local climate regulation	Above ground carbon sequestration	**T***
	Below ground carbon sequestration	**TD***

Deadwood is an important carbon sink. Assuming the alternative is the rapid combustion or decomposition of deadwood carbon stock, maintaining deadwood stocks in woodland is associated with a high confidence of increasing below ground carbon sequestration.

Current recommended densities for deadwood are $20m^3$ ha^{-1} by Forest Research, and suggested management practices to increase deadwood are 1. Allowing natural processes (e.g.. wind damage) to create deadwood, 2. Protecting existing deadwood stocks, 3. Diversify stand structure to encourage the creation of new deadwood, and 4. Link or buffer existing deadwood habitats (Humphrey & Bailey, 2012). The existing evidence is not sufficient to review these actions separately, however general comments on the role of deadwood in carbon sequestration are below.

Estimates from Matthews et al., in prep. put the average carbon stock of deadwood in woodlands in England at 11 tC ha^{-1} (4% of woodland carbon stocks). The percentage of woodland carbon stocks in deadwood across the four nations in the UK is similar, with variation largely determined by the size of woodland soil carbon stocks (Table 2). In woodland that is not clearfelled, carbon in deadwood can contribute a moderate carbon stock, of a similar magnitude to litter, representing approximately 10-15 tC ha^{-1} by 45 years of age (Matthews et al., in prep.). In clearfelled woodland, deadwood is a large component of retained carbon stocks, and Matthews et al., in prep. report deadwood carbon stocks of 119 tC ha^{-1} at the site of a Sitka spruce woodland on organo-mineral soil, 10 years into its second rotation, compared to 0.5 tC ha^{-1} mid way through the first rotation. It is likely that these values are an underestimate, due to the largely undocumented contribution of buried deadwood to carbon stocks (Moroni et al., 2015). As deadwood decomposes, some its carbon stock will contribute to the soil carbon pool. (Kahl et al., 2017) measured the decay rates of deadwood across 13 European tree species and found that the average decay rate constant for density loss (k) was significantly less for gymnosperms (0.014 y^{-1}) than for angiosperms (0.046 yr^{-1}), but that a range on enzyme activity signatures and environmental variation also effected decomposition rate. Although decomposing deadwood can release a large amount of DOC per unit mass, given its relative abundance compared to other sources of DOC (leaf litter, top soil etc.), there is evidence that deadwood contributes a relatively small amount of DOC losses overall (Hollands et al., 2022). Standing deadwood can also contribute to above ground carbon stocks, particularly in veteran and ancient trees (Read, 2000).

An alternative suggestion to the preservation of deadwood carbon stocks in woodland, is active preservation of deadwood carbon stocks in peat, which has been shown to inhibit decomposition and preserve exogenous carbon sources (Fenner & Freeman, 2020). This is likely to result in a higher percentage of carbon in deadwood remaining sequestered than if left in situ, but would not provide the additional benefits for biodiversity that are anticipated from increasing woodland deadwood stocks (Humphrey, 2005). This is not a management that has been practically tested, and would require further research prior to implementation.

Given the evidence base above, expert opinion suggests that the preservation of deadwood stocks is likely to be beneficial for carbon sequestration, assuming the alternative is more rapid loss of deadwood carbon stocks, thorough combustion or an alternative method of disposal. If this assumption is incorrect and deadwood carbon stock are preserved through some other means, as suggested by (Fenner & Freeman, 2020), benefits will be less clear.

Preserving existing woodland biomass to allow deadwood to develop and accumulate naturally is also likely to increase total carbon stocks (see 3.24.6).

Any measures that encourage the premature coppicing or felling of otherwise healthy and productive trees to enhance dead wood is likely to have a negative effect on carbon sequestration, by reducing rates of primary productivity and associated emissions with felling and potential soil damage (Matthews et al., in prep.). Increasing litter inputs has also been associated with the priming of soil respiration rates to increase rates of carbon loss from soils, due to the metabolism of old carbon stores (Sayer et al., 2011). However, rates of organic matter release to a soil are likely to be slow compares to litter.

3.24.5.2 Co-Benefits and Trade-offs

Deadwood can act as a valuable habitat for invertebrates, contributing to biodiversity (Humphrey & Bailey 2012).

3.24.5.3 Magnitude

In most UK woodland, deadwood likely contributes approximately 10-15 tC ha^{-1} to carbon stocks (Matthews et al., in prep.).

3.24.5.4 Timescale

Variable
A change in deadwood management to increase natural deadwood production may take many decades to have an effect, although other benefits for carbon sequestration will likely be achieved in the interim (see 3.24.6). Natural deadwood deposition also increases with age.

Reducing the removal of deadwood that does not pose a risk of disease transmission or a health hazard to people could impact deadwood stocks immediately.

3.24.5.5 Spatial Issues

No assessment.

3.24.5.6 Displacement

No assessment.

3.24.5.7 Maintenance and Longevity

Management to increase natural rates of deadwood deposition over time is likely to be a long-term requirement. A subsequent reversion in management could potentially result in the loss of deadwood stocks on site, and a reduced rate of production.

3.24.5.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.24.5.9 Climate Factors / Constraints

No assessment.

3.24.5.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.24.5.11 Uptake

Standing deadwood can be a health risk to the public in extreme cases, which is likely to take priority.
In other cases, deadwood can be perceived as a hazard to the tree and potentially increase the risk of tree fall.

3.24.5.12 Other Notes

None

3.24.6 EHAZ-138: Manage vegetation to reduce the risk of wildfires

Duplicated evidence: Create / maintain fire breaks to minimise the spread of wildfires [**ECAR-042**]

3.24.6.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**TD***
Global, regional & local climate regulation	Above ground carbon sequestration	**TD****
	Below ground carbon sequestration	**LTD***

Although woodland fires are relatively rare in the UK currently, their impact can be disproportionately high in terms of economic and environmental damage. The area affected by woodland fire in the UK each year is highly variable and the average area burned was approximately 2100 ha yr^{-1} (423 to 8675 ha yr^{-1}) in the UK and 233.3 ha yr^{-1} (29 to 979 ha yr^{-1}) in England from 2010 to 2017[8]. Forest fires are a growing risk in the UK as a result of climate change, primarily in the south and East of England but exact risk a are unknown (Arnell et al., 2021). The recovery of woodland carbon stocks following wildfire will be dependent on regeneration, but full recovery of carbon stocks could take many decades. Wildfire in ancient or near natural woodland could result in not only a large loss of carbon but also significant damage to biodiversity. The magnitude of carbon emissions attributable to woodland fires in the UK are unknown, and will vary with the intensity of the wildfire, in addition to the area affected.

Managing of vegetation to reduce the spread of wildfires could entail the construction of fire breaks or the removal of litter and highly flammable biomass to reduce fuel load, potentially by proscribed burning. Each of these managements will be associated with the removal of woodland biomass and carbon stock to varied degrees and will have secondary effects on biodiversity, soil health and air quality (Agra et al., 2012; Roberts & Wooster, 2021). There has been little empirical study regarding the effectiveness of fuel breaks or vegetation management for the overall emissions balance for reducing the damage done by forest fires. A study by (Syphard et al., 2011) assessed whether fuel breaks successfully halted fires in California over a period of 26 years, and found a 46% success rate, that was strongly associated to the presence of additional fire suppression activities. Access by firefighters, fire size, larger fuel breaks, younger vegetation and good maintenance of fuel breaks were significantly associated with fire containment. Khabarov et al. (2016) modelled the area affected by wildfire across several European countries over the next century and showed that prescribed burns significantly reduced the extent of forest wildfires under most climate projections. However, there analysis does not estimate the magnitude of carbon emissions across the multiple treatments and emissions from prescribed burns would need to be accounted for.

In the absence of further evidence, expert opinion suggests managing vegetation for wildfire is likely to be beneficial long term, due to the risk of large emissions from uncontrolled wildfire. However, the disruption to carbons stocks caused by management should be balanced with risk on a case-by-case basis.

3.24.6.2 Co-Benefits and Trade-offs

The removal of plant biomass (such as deadwood, see section 3.24.5) will affect the provision of other ecosystem services and potentially undermine the benefits to biodiversity from having that biomass present. The co-benefits of having this biomass present are discussed elsewhere with this review.

3.24.6.3 Magnitude

Unknown, likely to vary greatly in magnitude as a result of changes in long term fire risk and management implementation.

3.24.6.4 Timescale

>10 years
Greatest returns are likely to occur over long timescales, where there are risks of large fires.

3.24.6.5 Spatial Issues

Targeting proportional to risk is likely beneficial.

3.24.6.6 Displacement

No assessment.

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[8]www.forestresearch.gov.uk/tools-and-resources/statistics/forestry-statistics/forestry-statistics-2018/environment/woodland-fires/

3.24.6.7 Maintenance and Longevity

No assessment.

3.24.6.8 Climate Adaptation or Mitigation

Managing woodlands for a potential increase in wildfire risk under climate change would constitute climate change adaptation. Reducing the loss of sequestered carbon to the atmosphere as a result of wildfire would constitute climate change mitigation.

3.24.6.9 Climate Factors / Constraints

Targeting areas with the highest risk of wildfire due to climatic conditions is likely to be beneficial.

3.24.6.10 Benefits and Trade-offs to Farmer/Land-manager

Wildfire is associate with potentially significant risks to personal wellbeing, property and health, therefore managing wildfire in proportion to risk may have wider benefits for land owners.

3.24.6.11 Uptake

No assessment.

3.24.6.12 Other Notes

None

3.25 Restoration, management and enhancement – Woody features

3.25.1 EBHE-205EM: Enhance/ manage wood pasture (e.g. through appropriate grazing)

3.25.1.1 Causality

Global, regional & local climate regulation	Above ground carbon sequestration	**T****
	Below ground carbon sequestration	**LTD***

<u>Managing woody vegetation</u>
It is well established that woody vegetation can store above and below ground carbon (in both root biomass and soil carbon) long term, with biomass accumulating over the course of many years. For the impacts of managements such as coppicing and thinning, see section 3.24.

Trees growing in open locations have reduced competition above and below ground and have been shown to grow faster than densely packed trees. Upson et al. (2016) found that a silvo-pasture system with integrated trees and grassland stored 5% more carbon (above and below ground) than the equivalent area of trees and grassland grown separately, after 14 years. This was in part attributed to the greater size of trees grown in silvo-pasture. The optimal distance between trees for maximising carbon sequestration will vary between species and species combinations (Forrester et al., 2017), and may conflict with other management needs for the system. Whether difference will persist to greater stand ages is unclear (Gregg et al., 2021). Rapid initial sequestration rates will ultimately slow as stands and trees reach maturity and carbon stocks in vegetation biomass stabilise, subject to management (Matthews, 2020). Significant losses of soil carbon have been reported following the establishment of silvo-pasture trees, there minimising additional losses during management is likely to be beneficial, particularly in areas of high rainfall (Prosser et al., 2022). Woodland in silvo-pasture is frequently subject to coppicing or pollarding, particularly veteran tree (Kirby et al., 1995;

Read, 2000). There are few studies reporting their effect of coppicing or pollarding on overall carbon sequestration potential, although available evidence shows a reduced long-term average carbon stock with increased management intensity.

Grazing

The interactions between grazing and woodland pasture carbon stocks are complex, with evidence suggesting both intensive grazing and no grazing can be detrimental for soil carbon stocks in some cases, and that responses to a change in management are site dependent (Abdalla et al., 2018). The majority of evidence suggests that low intensity grazing will be beneficial for grassland carbon stocks. However, the emissions from livestock themselves should also be considered. Protecting young trees from grazing and browsing as a part of the regeneration of wood pasture will promote the long term sustainability of the system. Lack of regeneration in wood pasture is associated with over grazing (Kirby et al., 1995). Compaction and root damage are deleterious to woody vegetation, and can lead to symptoms of drought and nutrient deficits if severe (Kozlowski, 1999). Whilst there is not empirical evidence assessing the effects of avoiding compaction on rates of carbon sequestration and storage in trees specifically, there is a robust logic chain suggesting that this will potentially increase the ability of trees to sequester carbon and preserve existing carbon stocks, particularly in old trees.

3.25.1.2 Co-Benefits and Trade-offs

Tree cover will provide shade and a potentially improve livestock welfare where sun exposure is a concern.

3.25.1.3 Magnitude

Upson et al. (2016) fount that the overall effect of silvo-pasture (14 years after planting) was a significant reduction in soil carbon by 6.1 tC ha^{-1}, compared to control pasture systems, in the top 10cm of soil, but no significant differences below that depth. Carbon sequestered in above and below ground tree biomass in the silvo-pasture system was equivalent to -99.4 tC ha^{-1}.

The effects on carbon sequestration of specific managements in silvo-pasture and on sequestration by trees are unclear.

3.25.1.4 Timescale

Unclear, though most of the studies reviewed by Abdalla et al. (2018) in the responses of below ground carbon sequestration to grazing pressure cover timescales >10 years. Importantly, the gains in terms of SOC sequestration are readily reversible (Godde et al., 2020).

3.25.1.5 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.25.1.6 Other Notes

None

3.25.2 ECCM-056: Manage veteran and ancient trees

3.25.2.1 Causality

Global, regional & local climate regulation	Above ground carbon sequestration	**LTD****
	Below ground carbon sequestration	**LD***

There is limited published evidence for rates of carbon sequestration and size of carbon stocks in veteran and ancient trees in the UK or their importance at a national scale. However, there is a good logic chain suggesting these features are likely to store significant carbon at the individual level. As a result, managing these trees in a way that promotes their conservation is likely to help preserve existing carbon stocks.

Modern management of veteran trees is primarily intended to preserve trees for as long as possible or manage them for biodiversity. In many cases, active management may simply entail monitoring trees, and only intervening when necessary. The primary reasons for cutting ancient and veteran trees are for public safety, landscape and aesthetic value, and maintaining habitat conditions for biodiversity (Read, 2000).

Where management (e.g. pruning or pollarding) occurs to control disease or reduce the risk of tree death (e.g. due to falling) logic chains suggest this will promote the preservation of standing above and below ground biomass. However, there is good evidence that many veteran trees will respond badly to cutting, potentially leading to the death of the tree, and so sufficient care should be taken to minimise this risk, balancing against other trade-offs (Read, 2000). Where intensive management of ancient trees has continued, such as regular pollarding, reduced management intensity and frequency would likely result in an increased standing carbon stocks (Matthews, 2020). The removal of biomass itself may promote the generation of new growth, but pollards can also fail and any cutting could be reduced (Read, 2000). Ensuring that subsequent generations of veteran trees have a chance to develop and that stands of ancient trees can be sustainable long term will help preserve these carbon stocks. Matthews et al., in prep. identified one study of the carbon stocks in veteran trees, by (Hale et al., 2019), where trees had been unmanaged for more than 100 years.

Assessments of tree carbon stocks over 65 years, using repeat measurements of tree size and allometic scaling, suggested the carbon is being sequestered at a rate of -1.3 tC ha^{-1} yr^{-1}, which is lower than the rates of sequestration in naturally regenerating stands of ages 85-100 years old also reported by Matthews et al. (in prep.). Hale et al. (2019) estimated that C stocks in the older growth stands approximately doubled, from an average of 8.92 kg C m^{-2} in 1945 to 17.50 kg C m^{-2} in 2010.

3.25.2.2 Co-Benefits and Trade-offs

Biodiversity and cultural value.

3.25.2.3 Magnitude

(Hale et al., 2019) estimated that C stocks in tree biomass of unmanaged old growth stands was approximately of 8.92 kg C m^{-2} in 1945 and 17.50 kg C m^{-2} in 2010.

The effects of pollarding on overall carbon sequestration are unknown, but likely to be negative.

3.25.2.4 Timescale

Varied
Management for the ongoing survival of veteran and ancient trees is necessarily long term. However, the benefits of preventing the loss of these carbon stocks is instant.

3.25.2.5 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.25.2.6 Uptake

There are a range of legal obligations on those owning, managing or working on ancient trees (Read, 2000) which prohibit some activities and may provide an administrative barrier to some management. However, this legislation acts to reduce the likelihood of harmful management.

The cultural value of trees as features of the landscape may offset some motivations for a change in management.

3.25.2.7 Other Notes

n/a

3.26 SOIL MANAGEMENT AND PROTECTION

This bundle is focused on strategies of soil management and protection that enhance the carbon sequestration of agricultural soils in the form of soil organic carbon (SOC). Carbon losses from these soils in recent decades are estimated to be 0.4 g C kg^{-1} yr^{-1} in England and Wales from 1978 to 2003 (Bellamy et al., 2005) and between 0.12 and 0.23 g C kg^{-1} yr^{-1} from 1978 to 2007 across Great Britain, excluding Northern Ireland (Reynolds et al., 2013). Furthermore, simulated carbon losses over longer periods of time, including since the onset of industrialisation (Janes-Bassett et al., 2021) and over the last 12,000 years of human land-use since the Last Glacial Period ended (Sanderman et al., 2017) underscore the extent to which management of land for agriculture has impacted carbon storage within soil. Management and protection of soil to mitigate and even reverse losses of carbon has been highlighted as a priority in academic research for several years (Amelung et al., 2020; Dawson & Smith, 2007; Smith et al., 2000). Some sources argue that where there is significant potential for carbon sequestration is in cropland soils, particularly where yield gaps and/or historic losses of carbon are largest (Amelung et al., 2020). However, others have argued that the potential for carbon sequestration in agricultural soils is overstated (Berthelin et al, 2022; Janzen et al. 2022; Smith et al. 2005). A recent review of the greenhouse gas removal potential of carbon storage in mineral soils, through practices such as reduced tillage, cover crops and organic matter additions, was limited by a lack of scientific consensus in the impacts of the various practices but gave a central estimate of a sequestration potential of 4.7 tCO_2 ha^{-1} by 2050, or 15.7 Mt CO_2 yr^{-1} in 2050 in the UK, due to the relatively large area available for implementation (Element Energy & UKCEH, 2021). Of the actions proposed (all of which are discussed in more detail below), key findings include:

- Cover cropping, growing perennial crops, and incorporating ley rotations into arable systems have good potential for improving below ground carbon sequestration, mainly through increased inputs of organic C. Increased SOC normally leads to improved soil physical structure, which is beneficial in itself, but may also give positive feedback leading to additional C sequestration through stabilisation of organic matter in aggregates. This in turn can often facilitate improved root growth, leading to increased efficiency of use of nutrients and water. Improved structure also, importantly, leads to decreased soil erosion.
- Evidence for the effect of green manures is mixed, and more research is needed on the effects of green manures on soil carbon storage, especially in the UK context.
- Both optimising soil pH (primarily through liming of acidic soils) and reducing fertiliser use can yield benefits to soil carbon sequestration, but both these interventions need to be properly targeted to avoid significant trade-offs such as reduced land productivity and substantial GHG emissions, as discussed in more detail below for the respective actions.

Reducing tillage through minimum or even no-tillage cultivation can help to sequester carbon in soils. However, the effect of this particular intervention has been argued by some as being over-stated, with evidence that gains would be undone by any intermittent tillage, as required by broader management objectives.

3.27 SOIL MANAGEMENT AND PROTECTION – COMPACTION MANAGEMENT

3.27.1 EHAZ-028: Restrict the grazing season where there is a risk of causing soil compaction, run-off and erosion

Duplicate evidence base: Reduce grazing and stocking rates when soils are wet to avoid soil compaction [**ECPW-249**]; **ECPW-040**: Create/ maintain livestock tracks; **ECPW-003** Avoid cultivation and trafficking on wet soils; **ECPW-297** Minimise trafficking and manage land to reduce soil erosion and loss around field structures such as livestock shelters /feeders/ troughs: e.g. for outdoor pigs.

3.27.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**N**
	Below ground carbon sequestration	*

The effect of grazing on SOC is complex. Grazing is used to extract yield from grasslands; however, this also removes carbon from the system (Soussana et al., 2007), influences soil moisture and subsequent plant growth – and by extension, SOC . Eze et al. (2018) conducted a global meta- analysis of the effects of grazing intensity on soil carbon stocks and found a consistent and significant decrease in carbon stocks with increasing grazing pressure from no grazing to light (-6.9%), to moderate (-13.2%) and to heavy grazing (-27.1%). Their definition of levels of grazing intensity is not quantitative, and not all contributing studies may be informative to circumstances in the UK, and prior land use and state of degradation is not accounted for. For a further review of the effect of reducing grazing intensity of carbon sequestration see section 3.19.1.

Soil compaction problems can be ameliorated through grazing management improvements, including reductions to the grazing season where compaction, runoff and erosion risks are high. Soil compaction from heavy machinery and livestock grazing reduces the soil air volume, thereby increasing the water-filled pore space for the same moisture content as a less compacted soil, which reduces infiltration and promotes surface runoff and erosion (Kozlowski, 1999; van der Weerden et al., 2012). Rates of soil compaction increase with trampling intensity, soil moisture, plant cover, slope and land use, with the magnitude of compaction extending up to 5, 20 and 60 cm, depending on whether the compaction is associated with natural pressures, foot traffic or heavy machinery (Nawaz et al., 2012). Compaction affects the CO_2 concentration and mineralisation of SOC, with compaction noted to reduce the former in favour of stimulating CH_4 emissions via methanogenic bacteria under anaerobic conditions (Nawaz et al., 2012, and references therein).

Overgrazing can result in bare ground and soil erosion, particularly near river banks and water bodies where soils are wet (Alison et al., 2019). The prevention of bank poaching by excluding livestock will reduce the loss of soil carbon to river and pond systems through erosion, although current rates of erosion from back poaching are unclear (Bilotta et al., 2007; Pulley et al., 2021). Expert opinion suggests that local rates of soil erosion as a result of poaching could be significant. The fate of carbon that is lost from terrestrial to aquatic systems is also unclear. Particulate organic matter may be subsequently sequestered in other wetland, coastal or marine habitats, and soil nutrients could prompt an increase in productivity in some systems (Beaumont et al., 2014; John N. Quinton et al., 2010). Where vegetation is allowed to recover, there could be an increase in above ground biomass.

Grazing systems at a global aggregate scale currently emit more GHGs than they sequester in the form of SOC stocks. Although grazing-induced sequestration offers a climate mitigation strategy, it should be promoted by targeted farming strategies where permissible, and ultimately its global mitigation potential is lower than frequently suggested (Godde et al., 2020).

3.27.1.2 Co-Benefits and Trade-offs

No assessment.

3.27.1.3 Magnitude

The impacts of grazing pressure on soil carbon sequestration specifically as a result of soil compaction and erosion in England have not been estimated, as far as the authors are aware.
Abdalla et al. (2018) summarise the general impacts on SOC under different grazing pressures and climate conditions:

- ***Dry warm climate***: SOC declines under all conditions apart from low grazing intensity where SOC increases by 5.8%.
- ***Moist cool climate (which best characterises England)***: SOC declines (−19%) under all grazing intensities.
- ***Moist warm climate***: SOC increases (+7.6%) under all grazing intensities.
- ***Dry cool climate***: SOC increases (+16.1%) under low to medium grazing intensities; the effect of high grazing intensity is unknown.

Estimates of the effect that include reduced emissions from livestock directly suggest that reduced intensity of livestock management between 1991 and 2010 led to the sequestration of -1.5 gC m-2 yr-1 in the British Isles, separate from any associated changed in land cover (Chang et al., 2016).

3.27.1.4 Timescale

> 10 years
Recovery times in soil carbon stocks that have been degraded from compaction or erosion may be initially rapid in extreme cases, but can also take many decades to reach a new equilibrium (Smith, 2014).

3.27.1.5 Spatial Issues

The restriction of grazing pressure in areas with evidence of soil compaction, poaching and erosion or when soils are wet is likely to be broadly beneficial to below ground carbon sequestration.

3.27.1.6 Displacement

Reduced grazing, and preventing its expansion in the uplands, could lead to increased grazing elsewhere to keep up with demand (Alison et al., 2019). On the other hand, productivity gains may reduce land pressures and emissions per kilogram of milk and meat produced from grass-fed animals (Godde et al., 2020).

3.27.1.7 Maintenance and Longevity

Unclear if any gains in SOC would be maintained if grazing intensity re-intensifies to its previous state, though a return to overgrazing would most likely result in SOC losses (Alison et al., 2019).

3.27.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

Reductions in GHGs emitted from grazing livestock as a result of soil compaction could be realised if displacement is avoided through grazing management.

3.27.1.9 Climate Factors / Constraints

Not assessed

3.27.1.10 Benefits and Trade-offs to Farmer/Land-manager

Potentially improved land productivity if compaction, runoff and erosion are reduced and belowground biomass increases.

3.27.1.11 Uptake

Grazing is ultimately one of the ways in which farmers make money, so there will be economic barriers to interventions that restrict grazing. In upland areas, while incentives to reduce stocking rates may be readily taken up by some farmers, reducing stocking may run counter to some farmers' ideologies. Furthermore, hefting of sheep on common land may become harder over time with declines in sheep numbers, potentially resulting in a positive feedback and abandonment of some upland areas (Alison et al., 2019). Elsewhere, farmers may be open to incentives for rotational grazing, for instance to enhance plant productivity and by extension increase SOC (Alison et al., 2019).

3.27.1.12 Other Notes

More primary research is needed into determining the optimum grazing pressure on a range of grasslands spanning various configurations of soil type, plant species, grazing management practices and climate. It is not clear that there are consistent definitions of "light", "moderate" or "heavy/extreme" grazing in the literature. Normalised grazing intensity metrics based on regional or global datasets (*sensu* Abdalla et al., 2018) offer one potential avenue to pursue going forward.

3.27.2 ECPW-039: Aeration of soils in grassland situations to remove surface compaction / capping especially from sheep grazing

3.27.2.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**L***
	Below ground carbon sequestration	**LT***

Compacted soils can be recovered naturally through precipitations, wetting and drying cycles, subsequent soil cracking, freeze–thaw cycles, and bioturbation which includes earthworm burrowing and root penetration and decay (Nawaz et al., 2012). In the case of lightly compacted soils, repeated alternating dry and wet periods can ameliorate compaction of clay soils, though recovery for sandy soils is weaker (Nawaz et al., 2012). Mechanical methods such as deep ripping can help with aerating soils in croplands and grasslands, though mechanical work becomes difficult near trees due to the presence of stumps and large roots (Nawaz et al., 2012). However, addressing the cause of soil compaction is also necessary to prevent issues reoccurring.

The decreasing of soil compaction by aeration and its impacts on vegetation growth (and associated above- and below-ground carbon sequestration) is likely to vary with soil type, although evidence is for growth of woody species (Kozlowski, 1999). If aeration is successful at reducing soil compaction from levels which were impeding root growth, logic indicates there may be benefits for carbon sequestration and preservation. Indeed, reversing compaction should stem losses of SOC, though it should also reduce bulk density which makes it difficult to determine the overall effect on SOC stocks. However, other studies have shown changes in soil compaction from aeration to be short lived (Cournane et al., 2011). When combined with the fact that grazing management's climate mitigation potential is modest globally (Godde et al., 2020), it is unlikely that removing compacted soils with have anything more than a negligible impact on SOC sequestration in grasslands.

3.27.2.2 Co-Benefits and Trade-offs

3.27.2.3 Magnitude

Unknown.

3.27.2.4 Timescale

>10 years

Highly variable timeframe depending on how compacted the soil is. Nawaz et al. (2012) suggest an average of 5 to 18 years depending on the soil type, climate and degree of compaction. In extreme cases, full recovery time for a heavily compacted soil can range from 100 to 190 years. Rapid natural amelioration of physically deteriorated topsoil to about 5 cm is possible but below 15 cm natural rejuvenated process is very slow (Nawaz et al., 2012).

3.27.2.5 Spatial Issues

Natural recovery rates can vary by soil type, climate and degree of compaction (and hence ultimately intensity of grazing pressure).

3.27.2.6 Displacement

Increased sheep grazing pressures on uncapped land.

3.27.2.7 Maintenance and Longevity

Some studies have indicated that changes in soil compaction from aeration are short-lived (Cournane et al., 2011).

3.27.2.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation. However, overall mitigation is unlikely to be significant, particularly if uptake is low and given both the slow compaction recovery times and readily reversible nature of the interventions.

3.27.2.9 Climate Factors / Constraints

Compaction amelioration rates vary by degree of compaction, soil type and climate conditions.

3.27.2.10 Benefits and Trade-offs to Farmer/Land-manager

Unknown

3.27.2.11 Uptake

Unknown

3.27.2.12 Other Notes

More primary research is needed to determine the effectiveness of removing compacted soils on SOC sequestration.

3.27.3 **ECPW-003** Avoid cultivation and trafficking on wet soils

Duplicate evidence: **ECPW-255**: Reduce weight of field machinery; **EHAZ-017**: Use low ground pressure tyres; **ECPW-294** Create/ maintain machinery tracks; **ECPW-296** Minimise trafficking and manage land to reduce soil erosion and loss around field structures such as poly-tunnels, plastic sheeting/ cloches or irrigation equipment used for horticultural crops; **EHAZ-031**: Use controlled traffic farming (CTF).

3.27.3.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**N**
	Below ground carbon sequestration	**L***

According to Antille et al. (2015): “Controlled traffic farming (CTF) is a system whereby: (1) all machinery has the same or modular working and track width so that field traffic can be confined to the least possible area of permanent traffic lanes, (2) all machinery is capable of precise guidance along those permanent traffic lanes, and (3) the layout of the permanent traffic lanes is designed to optimise surface drainage and logistics”. In the absences of CTF, random traffic patterns emerge as a result of variable track widths and operating patterns, which can cover up to 85% of the field area each time a crop is produced (Antille et al., 2015). Fields where CTF has not been adopted can become particularly prone to compaction, which reduces the soil air volume, thereby increasing the water-filled pore space for the same moisture content as a less compacted soil, which reduces infiltration and promotes surface runoff and erosion (Kozlowski, 1999; van der Weerden et al., 2012). Compaction affects the CO_2 concentration and mineralisation of SOC, with compaction noted to reduce the former in favour of stimulating CH_4 emissions via methanogenic bacteria under anaerobic conditions (Nawaz et al., 2012 and references therein).

Implementing CTF increases soil porosity by 5 to 70%, quadruples water infiltration rates, and doubles saturated hydraulic conductivity (Antille et al., 2015). Combined with greater cropping opportunities and enhanced yields for given rainfall and fertiliser impact, coupled with reduced reliance on conventional tillage, CTF can enhance SOC sequestration.

3.27.3.2 Co-Benefits and Trade-offs

Facilitate adoption of minimum-tillage or no-tillage cultivation [**ETPW-092**].

[TOCB Report-3-5D Systems **ECPW-255**] Soil compaction is likely to have little effect on above-ground biodiversity, except via effects on soil-probing species, which may be affected by reduced earthworm abundance or accessibility (Wardle 1999), but this is likely to have a very limited spatial footprint and so low impact, and reducing it will have a similarly small effect.

3.27.3.3 Magnitude

Fuel savings noted by Antille et al. (2015) could lead to GHG emission reductions from farm machinery of 0.126 t $CO_2e\ ha^{-1}\ yr^{-1}$. CTF has the potential to sequester more carbon through increased crop yield and cropping efficiency – e.g. between 10 and 30% for winter cereal crops under UK climate conditions compared to non-CTF farming (Antille et al., 2015).

3.27.3.4 Timescale

Unclear in the UK context at present given the limited uptake of CTF. Timescales of 3-6 years have been noted for fields supporting dryland grain crops in Australia (Tullberg et al., 2018).

3.27.3.5 Spatial Issues

Unclear, though the returns in terms of SOC sequestration would likely depend on degree of soil compaction prior to adopting CTF, as well as climate, soil type and cropping or grazing patterns.

3.27.3.6 Displacement

Unclear, though improved cropping efficiency and crop yields would likely mitigate displacement effects of other interventions that may have opposite impacts on land productivity.

3.27.3.7 Maintenance and Longevity

Unclear, though reverting to random traffic patterns would likely lead to compaction and associated reductions in productivity and SOC storage.

3.27.3.8 Climate Adaptation or Mitigation

Adoption of controlled traffic farming should reduce GHG emissions from soil by 30–50% under dryland grain crops (Tullberg et al., 2018). Any net carbon sequestration constitutes climate change mitigation.

3.27.3.9 Climate Factors / Constraints

Unclear, though it is likely the magnitude of SOC sequestration under this practice will vary with climate, soil type and crop/land cover as per other interventions.

3.27.3.10 Benefits and Trade-offs to Farmer/Land-manager

Anecdotal evidence of farming becoming easier under CTF (Chamen, 2015). Improvements to soil structure should improve land productivity and yield. If adopted, the need for tillage will be reduced which could facilitate delivery of action: [**ETPW-092**] *Use minimum-tillage or no-tillage cultivation* (Chamen, 2015). Anecdotal evidence from farmers converting to CTF and minimum-/no-tillage suggests costs associated with tractor diesel fuel for field operations could be cut by half (Antille et al., 2015).

3.27.3.11 Uptake

In northern Europe, there has been a move to "subsidiarity" since the turn of the millennium - i.e. delegating extension responsibilities to farmer's groups and encouraging participatory involvement - which in England and Wales, produced the Agricultural and Horticultural Development Board (AHDB) to support research and knowledge transfer. Despite this, there has been relatively little uptake of CTF (partly through the inertia), though anecdotal evidence from the same review suggests that farmers who do adopt CTF have commented on how much easier farming becomes with this switch in practice (Chamen, 2015). The use of tramlines (a facet of CTF) has been more widely adopted.

3.27.3.12 Other Notes

More evidence is required in the UK context, with primary research focused on monitoring short-term (0-5 years) reductions of GHG emissions and on longer-term (>10 years) monitoring of SOC dynamics.

3.27.4 **ETPW-223: Assess soil structure and plan how to avoid and alleviate soil damage and compaction (soil management plan)**

3.27.4.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**N**
	Below ground carbon sequestration	**T**

High SOC sequestration via accumulation of SOM is favoured by management systems that add high inputs of biomass to soil, enhance activities and species diversity, strengthen mechanisms of nutrient cycling, minimise soil disturbance and improve soil structure (Lal, 2004). High SOM and associated SOC accumulation in turn enhances these aforementioned management system characteristics, including soil structure and its indicators (Blanco-Canqui et al., 2013). Common physico-chemical properties that are measured to assess soil structure include:

- SOM content (typically through loss on ignition in the laboratory)
- Bulk density, including degree of compactness (relative bulk density)
- Soil texture, including relative proportions of sand, silt & clay
- Aggregate stability
- Porosity
- Plant available water
- Hydraulic conductivity and infiltration
- Air permeability and air capacity
- Tensile strength/penetration resistance

Some of these properties can in turn be described further – e.g. aggregates can be characterised in terms of their size, porosity, shape, colour, ease of breakup, together with the identification of the presence of a tillage pan, depth of root penetration, or number of earthworms (Muñoz-Rojas, 2018).

A number of simple field-based methods, including the Visual Evaluation of Soil Structure (VESS) have been developed for simple assessment of soil structure (Franco et al., 2019). However, field methods are highly subjective and dependent upon operator experience (Rabot et al., 2018).

Laboratory-based methods allow for more quantitative assessments of soil structure, including particle size distribution (which relates to soil texture), loss on ignition to determine SOM (and estimate SOC concentration), aggregate stability, and bulk density (including its reciprocal, porosity, and its relative measure, the degree of compactness). Combined with repeated field campaigns, it is possible to analyse changes in soil structure and function over space and time (e.g. Reynolds et al. (2013) using the Countryside Survey of Great Britain). Smith et al. (2020) reviewed methods of monitoring soil carbon and verifying soil carbon change, and emphasise the need for repeat soil surveys, supported by measurement/monitoring, reporting and verification platforms which can provide benchmarking and subsequently facilitate national reporting and emissions trading.

SOM and/or SOC is often singled out as the main indicator to monitor, given that it is generally good for enhancing levels of soil structure, including attributes that are altered by soil compaction such as bulk density, porosity and aggregate stability (Blanco-Canqui et al., 2013; Prout et al., 2020). Recent benchmarking analysis using SOC/clay content ratios – with values of 1/8, 1/10 and 1/13 delineating the boundaries between “very good”, “good”, “moderate” and “degraded” levels of structural condition – revealed that 38.2 % of arable land in England and Wales is degraded, along with 6.6 % of improved grassland and 5.6 % of woodland soils (Prout et al., 2020). Such analysis is useful for targeting where SOC sequestration is most needed.

Reduced aggregate stability drives additional losses of SOC through increased susceptibility to runoff, interrill erosion and crusting (Rabot et al., 2018 and references therein). Laboratory measurements of aggregate stability can be coupled with various spatial thematic layers (e.g. aerial imagery, digital elevation models and land cover maps) to map soil compaction (Alaoui & Diserens, 2018). However, given the longevity of subsoil compaction in particular (Nawaz et al., 2012), knowledge of historical land use is needed to map soil compaction more effectively so that alleviation can be better targeted in the future (Alaoui & Diserens, 2018).

3.27.4.2 Co-Benefits and Trade-offs

No assessment

3.27.4.3 Magnitude

Unclear, and any impact will be dependent on appropriate actions being taken in response to assessments. It is suggested that the maximum technical potential SOC sequestration in mineral soils in 2050 for UK land area is approximately -15.7 Mt CO_2 eq yr^{-1} (Element Energy & UKCEH, 2021). Monitoring will be critical to efforts to achieving this potential.

3.27.4.4 Timescale

>10 years given the timescales required to alleviate soil damage and compaction naturally even before the returns of improved SOC storage are taken into account. SOC is typically assumed to saturate after 20-50 years, with some even suggesting beyond 100 years (Element Energy & UKCEH, 2021 and references therein).

3.27.4.5 Spatial Issues

A global meta-analysis of the application of the VESS method shows that temperate soils and sandy soils (regardless of climate zone) presented lower structural quality scores than any other soil types (Franco et al., 2019). Based on data for England and Wales, arable soils tended to exhibit lower SOC/clay ratios (indicating poorer soil structural quality) compared to woodland and improved grassland (Prout et al., 2020). As subsoil compaction can persist over decades, indicating a long memory effect (especially in the case of afforestation), knowledge of previous land uses must be taken into account for mapping areas subject to soil damage (Alaoui & Diserens, 2018).

3.27.4.6 Displacement

Unclear and likely to vary with type of intervention to alleviate soil damage and compaction. E.g. controlled traffic farming (CTF) could improve cropping efficiencies and facilitate adoption of minimum-/no-tillage, potentially reducing the likelihood of displacement effects. On the other hand, altering the grazing season to mitigate soil damage and compaction could displace grazing pressures and their effects elsewhere.

3.27.4.7 Maintenance and Longevity

Unclear in terms of impacts on SOC storage, though given the vulnerability of soils to lose SOC sequestration gains through management/land-use changes, it is recommended that assessment of soil structure and measures to alleviate soil damage and compaction should be maintained continuously.

3.27.4.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.
Magnitudes of effect are unclear and likely to vary with the measures taken to alleviate soil damage and compaction. Will also depend on the state of soil structure prior to alleviation.

3.27.4.9 Climate Factors / Constraints

Soils in temperate regions tend to score lower structural quality scores under VESS assessment than in tropical and subtropical regions (Franco et al., 2019).

3.27.4.10 Benefits and Trade-offs to Farmer/Land-manager

Ultimately, assessments of soil structure should lead to better targeting of the most appropriate and cost-effective measures to alleviate soil damage and compaction. If done well, the gains in land productivity and cropping efficiency could reduce costs and increase profits.

3.27.4.11 Uptake

Field assessments, such as VESS have become more widespread in their adoption in recent years, including in temperate regions (Franco et al., 2019). However, results of field assessments are subjective and highly dependent upon surveyor experience (Rabot et al., 2018). Increasing levels of soils survey data (including the GB-wide Countryside Survey becoming a rolling annual survey) could facilitate the delivery of benchmarking assessments (*sensu* Prout et al., 2020) which individual land-managers can compare their own soils against.

3.27.4.12 Other Notes

None

3.28 Soil management and protection – Cover cropping

3.28.1 ECPW-002: Minimise bare soil to reduce soil loss e.g. cover crops, crop residues, trees coppice etc.

Duplicate evidence base: Use restorative vegetation cover following destoning or lifting of root crops [**ECPW-005**]; Use of cover crops as an alternative to plastic mulch - Soil-enriching cover crops may be grown over the winter in the same beds where a food crop is to be planted the following spring and used in place as mulch [**ECPW-279**]; Ensure persistent continuous vegetation cover on land [**ECAR-044**]; Maintain soil cover (e.g. grass, crop or geotextile), to reduce soil erosion and loss around field structures such as poly-tunnels, plastic sheeting /cloches or irrigation equipment used for horticultural crops [**ECPW-095**]; Maintain soil cover (e.g. grass, crop or geotextile), to reduce soil erosion and loss around livestock shelters/feeders/troughs; e.g. for outdoor pigs. [**ECPW-295**]; Use under and over sowing [**EHAZ-004**]; Re-seed grassland by slot-seeding or over-seeding [**ETPW-270**]; **ECPW-025**: Harvest and establish the following crop early in the Autumn; Diversify arable rotations (including cover and catch crops, over and under sowing). [**ECCM-001**]; **ECPW-264.** Leave unharvested cereal headlands; **ETPW-229** Enhanced overwinter stubble.

3.28.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	N
Global, regional & local climate regulation	Above ground carbon sequestration	N
	Below ground carbon sequestration	*

Minimising the extent of bare soil, primarily through the establishment of cover crops is likely to prevent losses of SOC via erosion and may increase SOC through residue returns (Alison et al., 2019). The literature is replete with evidence from temperate regions demonstrating the effectiveness of cover crops in promoting SOC accumulation in soil.

Cover crops are fast maturing crops grown within a rotation following harvest to maintain soil cover during fallow periods and are often ploughed under as green manure or killed off by herbicides in no-tillage systems (Maskell et al., 2019). Cover crops increase SOC by increasing the influx of carbon to soils (particularly if plant residues are returned to the soil) and by preventing erosion losses (Moxley et al., 2014). Bare soil tends to lose carbon partly through erosion, though in the UK, erosion may represent a net carbon sink (Quinton et al., 2006). Cover crops may alleviate some of the effects of soil compaction, reducing SOC losses via erosion further (Nawaz et al., 2012).

Perennial plants typically increase SOC stocks at a faster rate than annuals, but the SOC accumulation rate varies considerably with region, species, and management context (Guo & Gifford, 2002). Leguminous crops may be more effective than non-leguminous crops for SOC sequestration (Bai et al., 2019) though some studies show no significant difference between the two crop types (Poeplau & Don, 2015).

Increased emissions of N_2O is noted as a risk, though there is a lack of consensus in the published literature on this issue. If N-fixing cover crops do increase C sequestration in agricultural soils, they may also lead to increased N_2O emissions (Lugato et al., 2018). Emissions of N_2O from terrestrial ecosystems are a function of available mineral N, soil water content, the availability of electron donors (such as labile C), and soil physical properties (Basche et al., 2014). Increasing the proportion of grain legumes/pulses grown in rotations is one of the most effective ways of promoting nitrogen use efficiency and reducing N_2O emissions from agriculture through enhancements in soil quality and reductions in manufactured N fertilisers (Maskell et al., 2019). The mulching effect of cover crop residues on the soil surface may increase soil water and the potential for denitrification depending upon timing of precipitation (Dabney, 1998). Additionally, decomposing cover crop residues can temporarily immobilise soil N and then later increase soil pools of labile C and inorganic N (Kaspar & Singer, 2011; Steenworth & Belina, 2008), which will also impact dynamics of N_2O emissions (Basche et al., 2014).

Although there is some uncertainty about how much cover cropping can contribute to carbon sequestration, the consensus is that this practice could be significant. Indeed, according to a recent global meta-analysis, if 15% of current global cropland were to adopt cover crops, it is estimated there would be 0.16 ± 0.06 Pg of carbon sequestered per year, which is ~1–2% of current fossil fuels emissions (Jian et al., 2020).

3.28.1.2 Co-Benefits and Trade-offs

[TOCB Report-3-5D Systems **ECPW-002**] This has not been reviewed for biodiversity as it is too general: the various options will have a range of negative and positive effects for different species, which will be variable across action-taxon combinations.

[TOCB Report-3-5D Systems **ECPW-005**] This is assumed probably to replace imminent ploughing and/or winter crops, so will have negligible effect on biodiversity.

[TOCB Report-3-5D Systems **ECPW-025**] Assuming that this applies to crops that are already winter-sown, this will have small negatives from destroying a short-lived crop-stubble a little earlier. If this replaces a spring crop, it will have very large negatives, as spring cropping, especially following a fallow stubble over-winter, provides critical, high-quality habitat for many priority farmland species.

3.28.1.3 Magnitude

Bai et al. (2019) conducted a global meta-analysis of the effect of smart agricultural practices on soil carbon content and reported that cover crops increase SOC by approximately 6%, relative to paired controls where cover crops were not used. Other meta-analyses have demonstrated increases in SOC over non-cover crop control groups of 0.32 t C ha^{-1} yr^{-1} (Poeplau et al., 2018) and 1.11 t C ha^{-1} yr^{-1} (McClelland et al., 2021). An annual rate of SOC increase of 0.32 ± 0.08 t ha^{-1} yr^{-1} at a mean soil depth of 22 cm was observed over 54 years where cover crops were incorporated (Poeplau & Don, 2015). However, much of the data underpinning these estimates are from data sources outside of the UK and following varied practices that will not always be applicable to the use of cover crops in the UK.

3.28.1.4 Timescale

5-10 years, though the magnitude of the increase in soil carbon with cover cropping increased with study duration (Bai et al., 2019).
Bai et al. (2019) measured significant effects of cover crops on SOC storage after 5 years, though it is unclear how long this magnitude may continue to increase in the future, with some of the studies cited in our report spanning multiple decades of observations (e.g. Poeplau & Don, 2015).

3.28.1.5 Spatial Issues

Cover crops can increase crop P nutrition and may be more effective in croplands with low available P (Hallama et al., 2019).

3.28.1.6 Displacement

Unknown.

3.28.1.7 Maintenance and Longevity

Management decisions interact with environmental factors to influence the maintenance of cover crops and the percentage response of SOC stock storage relative to a no cover crop system (McClelland et al., 2021). Longevity of continued SOC accumulation is unclear, though SOC would likely be lost if conditions reverted to the previous management regime.

3.28.1.8 Climate Adaptation or Mitigation

SOC sequestration and retention via minimisation of bare soil extent will likely contribute towards climate mitigation. There are risks of N_2O emissions increases from N-fixing cover crops, though it is unclear how much this might be balanced by reductions in artificial N fertiliser manufacture (Maskell et al., 2019). Some modelling suggests that over time, N_2O emissions from N-fixing cover crops could be great enough to counteract the climate mitigation effects of SOC sequestration (Lugato et al., 2014).

3.28.1.9 Climate Factors / Constraints

Soil and weather conditions at the time of establishment (and the time of sowing itself) are important determinants of the effectiveness of cover crops (Rollett & Williams, 2020). SOC impacts of cover crops are reported to be strongest in warmer regions (Bai et al., 2019). The evidence is mixed as to whether planting leguminous crops is more beneficial to SOC accumulation than planting non-leguminous crops, with Bai et al. (2019) finding the former to be more effective, while others have found no statistically significant difference between the two crop types (Poeplau & Don, 2015). Some consideration needs to be paid to ensure that cover crops do not act as a bridge for pests and disease (Gregg et al., 2021).

3.28.1.10 Benefits and Trade-offs to Farmer/Land-manager

If cover crops do increase crop P nutrition, this would likely increase yield (Hallama et al., 2019). Cover crops may reduce the required application rate of synthetic N fertilisers (Maskell et al., 2019). Implementing cover cropping requires an additional workload that may not be acceptable to all farmers.

3.28.1.11 Uptake

Cover crops come at an immediate expense to farmers in terms of seed and machinery costs (Alison et al., 2019). Uptake will be limited to farms with compatible crop rotations, due to restrictions on sowing time, and in many cases may only be applicable on multi-annual time scales.

3.28.1.12 Other Notes

None

3.28.2 ECCM-071: Use intercropping

Duplicate evidence base:
Diversify arable rotations (including cover and catch crops, over and under sowing). [**ECCM-001**]

3.28.2.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**N**
	Below ground carbon sequestration	**LTD***

Most evidence we have for the effects of intercropping on below ground carbon stocks is from tropical and Mediterranean climates, with relatively little evidence for temperate regions or the UK. Where there is evidence for the UK, much of it relates to agroforestry, for which the evidence is reviewed elsewhere in this report (see section 3.6).

Intercropping is the practise of growing two or more crops in close proximity to one another. It is thought that intercropping increases sequestration of SOC through increases in aboveground productivity brought about by species complementarity, as well as increased belowground productivity leading to greater inputs of root litter (Cong et al., 2015). However, negative interactions are also possible outside of well-established crop pairings. The intercropping of legumes, such as peas or soybean, with the primary cash crop (for example oats or maize) has more support based on logic chains and published evidence, due to the biological nitrogen fixation provided by legumes (Dyer et al., 2012). Intercropping can include cover cropping when cover crops are grown with the arable crop rather than during bare fallow periods as part of the arable rotation (Maskell et al., 2019).

When two or more arable crops are combined, some evidence suggests that SOC concentrations increase relative to monocrop fields. SOC concentrations were significantly greater in the maize-soybean intercrop fields than the soybean monocrop fields grown in the Argentine Pampa (Dyer et al., 2012). Soil CO_2 emissions were much higher from maize monocrop fields than either the intercrop or soybean monocrop fields, with negligible impacts on N_2O emissions (Dyer et al., 2012). A meta-analysis of Mediterranean field studies suggests that the use of intercropping, conservation tillage and organic fertilisation led to improvements in soil quality and fertility and maintenance of a ground cover that can protect soil (Morugán-Coronado et al., 2020). However, generalisation of these principles to other crop combinations and farming methods is not suitable. Further evidence is required to support the widespread implementation of intercropping, particularly over other interventions.

3.28.2.2 Co-Benefits and Trade-offs

Not assessed

3.28.2.3 Magnitude

There is very little evidence from a UK context, but studies from the tropics and some temperate regions provide potential figures/ranges. Conversion from pasture to agroforestry increased SOC stocks by 9 to 10% (De Stefano & Jacobson, 2018). In northern China, SOC content in the top 20 cm was 4 ± 1% greater in intercrops than in sole crops, indicating a difference in C sequestration rate between intercrop and sole crop systems of 184 ± 86 kg C ha^{-1} yr^{-1} (Cong et al., 2015).

3.28.2.4 Timescale

Unknown

3.28.2.5 Spatial Issues

Differing magnitudes of SOC sequestration and emission rate reductions depending on the combination of crops in question. In Mediterranean fruit orchards, the highest response in SOC was achieved by the growth of permanent crops in the alleys (Morugán-Coronado et al., 2020). Ultimately, every farm, climate and soil type will differ in terms of its suitability for various crop combinations.

3.28.2.6 Displacement

Unknown, though any local decline in agricultural productivity could contribute to intensification elsewhere (Alison et al., 2019).

3.28.2.7 Maintenance and Longevity

Difficult to state for a UK context. Hombegowda et al. (2020) noted that intensive pruning of hedgerow plants for 5 years yielded SOC concentration conservation efficiencies of 42 to 47%.

3.28.2.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.
If intercropping results in similar reductions in bare soil exposure and losses via erosion to cover cropping, then the SOC sequestration increases could be significant.

3.28.2.9 Climate Factors / Constraints

Unknown in the UK context, though in the Mediterranean, possible negative effects on SOC storage occurred in warmer and drier regions (Morugán-Coronado et al., 2020).

3.28.2.10 Benefits and Trade-offs to Farmer/Land-manager

Potentially improved yields if intercropping confers similar decreases in nutrient leaching to cover cropping.

3.28.2.11 Uptake

Herbicide management to keep control of weeds (e.g. black grass in wheat and barley fields). Adequate machinery to plant and subsequently to filter harvested crops is another barrier to uptake, that may be addressed by access to more advanced harvesting machinery.

3.28.2.12 Other Notes

Data from temperate regions and the UK are limited; more primary research is necessary. Few meta-analyses of the impacts of intercropping on SOC in general exist to our knowledge also.

3.29 Soil management and protection– Fertiliser, nutrient, manure and mulch management

3.29.1 ECAR-015: Replace nitrogen fertiliser application by using clover in pasture or arable cropping systems

Duplicate evidence base: Diversify arable rotations (including cover and catch crops, over and under sowing). [ECCM-001]

3.29.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	L*
Global, regional & local climate regulation	Above ground carbon sequestration	N
	Below ground carbon sequestration	LT*

Positive effects of biological nitrogen fixation and deep-rooted perennial plants such as clover are widely documented, though perhaps not in the England/UK context. Evidence comes from mixed sources; some where a few additional species have been incorporated into an improved grassland, while other evidence is derived from less-fertile, semi-natural species rich swards under different management practices (Maskell et al., 2019). Where the use of clover and other management interventions to enhance sward diversity have been undertaken to reduce manufactured fertiliser inputs, additional co-benefits including SOC sequestration have been achieved (Alison et al., 2019). The effectiveness of measures is likely to vary depending on whether biological nitrogen fixation is being utilised through cover cropping, intercropping, through crop rotation or through sward diversification.

In theory, diverse swards could increase SOC in grasslands via increasing yields, and if establishing deeper rooting plant species, carbon may be sequestered more securely into subsoils and promote the shifting of organic matter into the subsoil via pedoturbation and greater rates of associated bioturbation (Maskell et al., 2019, Garnett et al. 2017; Newell Price et al., 2019). In practice, restoring or enhancing plant diversity on species rich neutral grasslands has been shown to yield benefits in terms of increased soil carbon storage (Gregg et al., 2021). Part of the reason for this increased SOC storage is the presence of key functional groups that are complimentary to one another and beneficial for SOC (Alison et al., 2019). One study suggested that the presence of C4 grasses and legumes considerably enhanced SOC accumulation, by 193 % and 522 %, respectively (Fornara & Tilman, 2008). Legumes increase SOC via N fixation, increased root biomass accumulation (including in the subsoil >30 cm deep) and enhanced productivity (Mortenson et al., 2004). Belowground inputs of carbon generally contribute more to total SOC than aboveground inputs, with deep-rooting legumes such as clover helping to enhance SOC stocks at depth (Chenu et al., 2019).

Some evidence of enhancing sward diversity to reduce nitrogen fertiliser requirements and accumulate SOC suggests there are context-dependent outcomes (Alison et al., 2019). While increasing plant diversity can enhance carbon inputs to the soil microbial community (Steinbeiss et al., 2008), renovating and reseeding grassland to establish clover and other perennials can also trigger losses of SOC (Schilis et al., 2005). Crucially, if incorporating legumes such as clover leads to an increase in productivity without a concomitant increase in nitrogen fertiliser use, then genuine carbon sequestration into the soil is achievable (Maskell et al., 2019). Yet, while increases in SOC with sward diversity have been found (e.g. Fornara & Tilman, 2008), increases in the total potential for SOC accumulation are often unclear. It has been shown that grasslands cannot continue sequestering carbon in perpetuity (Smith, 2014).

Where the introduction of legumes such as clover to swards does reduce nitrogen fertiliser use, and soil and climate conditions are optimal, on site N_2O emissions could also be reduced and emissions associated with the production of inorganic fertilisers avoided (Maskell et al., 2019). Conversely, where legumes are introduced into swards not previously dependent upon nitrogen fertiliser, or where climate conditions are not optimal, the extra fixed nitrogen can induce net emissions of N_2O, especially in wet regions or on poorly draining waterlogged soils (Henderson et al., 2015).

Where the incorporation of legumes enhances grassland productivity, this will enable higher stocking densities and with that, methane emission that could offset as much as 26 % of the global net SOC sequestration potential of legume sowing (Henderson et al., 2015).

3.29.1.2 Co-Benefits and Trade-offs

Bai et al. (2019) found that SOC sequestration was greater under leguminous crops (including clover) than non-leguminous crops.

3.29.1.3 Magnitude

Variable. One meta-analysis reports that sowing legumes on grasslands can increase SOC by 0.75 t C ha^{-1} yr^{-1} (Conant et al., 2001). In the approximately 10 % of pasturelands where carbon sequestration rates in soils exceeds N_2O emissions, legumes could sequester carbon at a rate of -0.5 t C ha^{-1} yr^{-1} (Henderson et al., 2015).

The more modest rates of potential carbon sequestration in soils suggest -0.22 t C ha^{-1} yr^{-1} in temperate climates (Smith et al., 2008).

3.29.1.4 Timescale

>10 years
Although the rate of SOC accumulation on newly seeded grasslands increased over 4 years (Steinbeiss et al., 2008), it has been argued that detectable changes in SOC stocks typically occur >10 years (Maskell et al., 2019). Reductions in GHG emissions through the introduction of legumes like clover could occur within the first year of implementation, especially if thee substitute the use of manufactured fertiliser (Maskell et al., 2019).

3.29.1.5 Spatial Issues

Carbon sequestration in soils due to sowing of legumes including clover will exceed N_2O emissions on approximately 10 % of pasturelands (Henderson et al., 2015). Going forward, establishing diverse swards should be targeted towards better drained soils where improvements in productivity are likely to exceed N_2O emissions (Maskell et al., 2019).

3.29.1.6 Displacement

Potentially if introducing clover leads to reductions in stocking densities locally and thus, increased stocking densities elsewhere (Bullock et al., 2011).

3.29.1.7 Maintenance and Longevity

A diverse sward can be quite ephemeral, so frequent reseeding may be necessary to re-establish the desired sward (Maskell et al., 2019). It should be noted that reseeding and tillage counteract increases in SOC (Schilis et al., 2005). Further, there is uncertainty around the persistence of SOC increases following cessation of sward management (Alison et al., 2019).

3.29.1.8 Climate Adaptation or Mitigation

Potentially modest climate mitigation effects from carbon sequestration if done well. However, N_2O emissions can be a consequence of adding legumes such as clover to a pasture or arable cropping system, with the carbon sequestration effect of legume establishment exceeding N_2O emissions in only 10 % of cases (Henderson et al., 2015). Where the incorporation of legumes enhances grassland productivity, this will enable higher stocking densities and with that, methane emission that could offset as much as 26 % of the global net SOC sequestration potential of legume sowing (Henderson et al., 2015).
Mixed effects have been noted in relation to flood risk. Deeper rooting plants such as clover should improve infiltration (Weisser et al., 2017). However, reseeding operations on sloping land would lead to greater runoff and flood risk during the reseeding phase itself, especially if more soil is exposed to raindrop impact and surface runoff (Maskell et al., 2019).

3.29.1.9 Climate Factors / Constraints

Alison et al. (2019) suggest that seed mixes will likely need to be tailored to local climate conditions.

3.29.1.10 Benefits and Trade-offs to Farmer/Land-manager

There are mixed effects on yield: in some cases there could be a reduction in yield, although there could also be increases with increased plant species richness, as well as benefits from improvements in nutritional quality (Maskell et al., 2019). If implemented well, legume sowing could lead to reductions in fertiliser and pesticide use, saving farmers/land-managers money. It has also been noted that some legume species including white clover offer opportunities for improving animal health with less medication, due to the presence of bioactive secondary metabolites, while also improving yields such as milk from dairy cows (Lüscher et al., 2014).

Potential problems faced by land managers include the 1) maintenance of diverse swards long-term, requiring careful management, and 2) a potential increase in the incidence of frothy bloat (tympanites) in livestock requiring additional management to mitigate risk, although the use of legumes with high tannin and flavenoid content have also been shown to reduce the incidence of bloat hazard (Lüscher et al., 2014; Phelan et al., 2015; Rochon et al., 2004).

3.29.1.11 Uptake

There are costs associated with seeds, as well as establishing and maintaining diverse swards with the incorporation of clover (Maskell et al., 2019). However, there may be some appetite among farmers for financial incentives to introduce clover, with such practices reportedly being undertaken on some Welsh pastures, and in some cases the infrastructure for managing clover-incorporated swards likely already exists (Alison et al., 2019).

3.29.1.12 Other Notes

Enhanced sward diversity may sequester more carbon into the soil, especially when transitioning from poor to optimal management or on previously degraded soils. However, it may not be possible for sward diversity improvements to sequester carbon at as high a rate in the long-term, so measurements of SOC should be taken over an extended time period, e.g. in 5-year intervals (Maskell et al., 2019). Deeper rooted plant species such as clover and increases in net primary productivity may increase soil organic matter and move it at depth, so SOC should be monitored in both the top and subsoil (Ward et al., 2016). More primary research needs to be directed towards these types of measurements listed.

3.29.2 ECCM-023: Use green manures within the rotation

Duplicate evidence: Use mulches and organic matter to increase the water retention capacity of soil [**EHAZ-113**]; **ETPW-221** Add organic matter (e.g. paper pulp or sawdust waste) to soil.

3.29.2.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**N**
	Below ground carbon sequestration	**LTD***

Green manures include crop residues that are returned to the soil, cover crops that are grown to provide organic inputs, and organic fertilisers that are added directly to soil (Rollett & Williams, 2020). Green manures can increase the accumulation of SOC on cropland (Powlson et al., 2012) as a direct result of the addition of organic matter and increased plant productivity (including further residue production) (Maskell et al., 2019). Outside of agricultural systems organic matter inputs can induce a priming effect on microbial activity, leading to mineralisation of soil carbon and emissions of CO_2, making the overall effect on below ground carbon sequestration difficult to anticipate (Sayer et al., 2011; Zhu et al., 2017), however global reviews suggests rates of carbon loss due to soil priming are relatively small and unlikely to singificantly impact sequestration rates long term (Bastida et al. 2019). A global meta-analysis of 74 studies comparing below ground carbon sequestration from organic farming and non-organic farming found that organically farmed soils had significantly higher SOC concentrations by 0.18%, equivalent to an increase in below ground carbon stocks by 1.08 tC $ha^{-1,}$ and a change in net carbon balance by -0.45 tC $ha^{-1} yr^{-1}$ (Gattinger et al., 2012). However, the conclusions of Gattinger et al. (2012) have been criticised, as the organic systems in question received substantially higher C inputs from external systems than the conventional farming systems and the selection of atypical conventional farming systems (Leifeld et al., 2013). This critique also highlights that increases in local carbon storage due to the transport of manure from elsewhere does not constitute additional carbon sequestration (Leifeld et al., 2013). By focusing on local carbon stocks, the analysis by Gattinger et al. (2012) also excludes the wider impacts of changes in farming strategy (such as the potential expansion of agricultural land where organic yields are lower than under conventional farming) and does not account for the life cycle emissions associated with organic or inorganic input production.

The best choice of organic matter addition to directly lead to carbon sequestration is also unclear. Chenu et al. (2019) highlight several studies that demonstrate greater long-term SOC storage from labile, readily degradable compounds than from lignin-rich material. Part of this enhanced SOC storage may be explained by the fact that labile compounds are processed with greater carbon use efficiency by microbes, increasing microbial biomass; a second reason could be the migration of soluble compounds in soil between mineral surfaces where they can become protected (Maskell et al., 2019). However, it remains to be seen whether "fresh" or "processed" organic matter inputs will confer the greatest benefits to SOC accumulation (Maskell et al., 2019). Furthermore, accumulated carbon stocks will only constitute net carbon sequestration if the emissions associated with the life cycle of green manure's production is taken into account (Smith et al., 2008).

3.29.2.2 Co-Benefits and Trade-offs

[TOCB Report-3-5D Systems **ECCM-023**] Assuming that refers to the use of green manures instead of mineral nitrogen, phosphorus and potassium, this is likely to have some positive effects on soil biodiversity and dependent species, but there is an absence of evidence.

[TOCB Report-3-5D Systems **EHAZ-113**] We would expect a negative effect on vegetation if fertility is increased (Tonn et al. 2010). The research in this area is on soil function and biodiversity only, and is generally confounded with organic management.

3.29.2.3 Magnitude

Evidence appears to suggest that manure additions, including green manures, enhance carbon storage per unit of nitrogen compared with manufactured fertiliser(Moxley et al., 2014). In a review to inform the LULUCF inventory, estimated changes in SOC stocks induced by manure application to croplands range between 5 and 18 t C ha^{-1} (Moxley et al., 2014).
Manure inputs such as sewage sludge and green compost/manures had greater positive effects on SOC storage than from the maximum permitted application of biosolids to UK land, which increased SOC by 0.63 t C ha^{-1} yr^{-1} (Powlson et al., 2012). In one meta-analysis, cover crops, including those grown as green manure material, increased stored SOC by approximately 6 % (Bai et al., 2019). However, some studies have suggested that SOC storage efficiency from cereal residues is often lesser than from other biosolids (Wuest & Gollany, 2013).

3.29.2.4 Timescale

0-5 years
One study found that soil loosening and straw slurry incorporation into arable soil increased SOC by >20 g kg^{-1} after one year, with additional benefits on grain yield (Getahun et al., 2018). Increases in SOC via organic inputs were found in another study to be greatest within the first 20 years of application, beyond which they diminished (Powlson et al., 2012).

3.29.2.5 Spatial Issues

One study suggests that SOC inputs from cereal residues hinge on suitable amounts of nitrogen, phosphorous and sulphur (Kirkby et al., 2013). The amount of nutrients needed may be quite predictable, and if not already present might be supplemented using fertilisers (Alison et al., 2019). However, incorporating cereal residues may involve a strategy of trying to fertilise stubble, which is discouraged due to likely nitrate leaching (Moxley et al., 2014). Nitrogen addition needs to be carefully targeted to minimise trade-offs, with Smith et al. (2008) highlighting that improved agronomy and nutrient management consistently increased CO_2 mitigation potential.

3.29.2.6 Displacement

A key issue is the alternative fate of organic inputs such as cereal straw. If the straw is burnt in the field, it is preferable to incorporate the carbon into the soil (Powlson et al., 2011). On the other hand, burning cereal straw as a source of bioenergy could reduce fossil fuel combustion and help mitigate climate change (Powlson et al., 2008). Alternatively, straw could be used as animal bedding which would ultimately cause SOC to be incorporated elsewhere after it becomes part of farmyard manure (Powlson et al., 2011).

Alison et al. (2019) also highlights that green manure application on arable land must be linked with GHG emissions from livestock elsewhere, or to an increase in NPK fertiliser application.

3.29.2.7 Maintenance and Longevity

Unclear whether increased SOC would persist after organic inputs stop. The impacts on SOC accumulation are typically saturating with time (Powlson et al., 2012).

3.29.2.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation. However, increases in local carbon storage due to the transport of manure from elsewhere does not constitute additional carbon sequestration at the landscape scale, although there maybe local benefits to soil function.

Modelling scenarios for integrated crop residues and low soil disturbance indicate a new SOC equilibrium for Europe will be formed around 2050; thereafter, N_2O emissions began to increase systematically, offsetting earlier GHG emission reductions by 2100 (Lugato et al., 2014).

3.29.2.9 Climate Factors / Constraints

Soil and weather conditions at the time of establishment are important determinants of the effectiveness of cover crops grown for green manure material (Rollett & Williams, 2020). SOC impacts of cover crops in general are reported to be strongest in warmer regions (Bai et al., 2019). Some consideration needs to be paid to ensure that cover crops grown for green manure do not act as a bridge for pests and disease (Gregg et al., 2021).

3.29.2.10 Benefits and Trade-offs to Farmer/Land-manager

Green manures may reduce the required application rate of synthetic N fertilisers (Maskell et al., 2019).

3.29.2.11 Uptake

Redistribution of manure towards arable regions comes with severe barriers in terms of storage and transportation (Alison et al., 2019). The application of green manures is practical where supply is plentiful and the costs are likely to be minimal, with some potential cost saving is manufactured fertiliser use falls (Rollett & Williams, 2020).

3.29.2.12 Other Notes

More targeted research is needed for green manure impacts on SOC specifically. In previous reviews (e.g. Alison et al., 2019; Maskell et al., 2019), the topic of green manure has been considered with those of biosolids, compost and biochar when outlining their utility as a soil protection and management strategy to promote SOC storage.

3.29.3 ETPW-242: Reduce fertiliser (organic and inorganic) application to below conventional levels

Duplicate Evidence: Reduce fertiliser (organic and inorganic) applications in high risk areas [**ECPW-110**]; Use very low inputs on permanent grassland [**ECPW-171**];– reduce fertiliser / low input system; Manage a decline in soil nutrient levels for habitats / species that need low fertility [**ETPW-241**]; Use no fertiliser [**ECPW-173**]

3.29.3.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T****
Global, regional & local climate regulation	Above ground carbon sequestration	**N**
	Below ground carbon sequestration	**LTD***

There is mixed evidence with regard to using either more or less fertiliser on SOC uptake. Evidence from outside the UK supports an increase in SOC following fertiliser application to improved land on mineral soils (Alison et al., 2019 and references therein). Conversely, increasing SOC through reducing fertiliser application below conventional levels can be achieved through adoption of a plethora of other interventions reviewed here in this Soil Protection and Management bundle. Evidence for using no fertiliser altogether is even more scant.

Plant growth can be limited by the availability of nitrogen, phosphorous or potassium or sulphur or micronutrients, and manufactured fertilisers are applied with the intention of lifting these nutrient limitations and increasing plant productivity. Manufactured fertiliser can increase C inputs to the soil through plant material root exudates, and through the microbial community (Alison et al., 2019). Nitrogen fertilisation was found to have no significant effect on carbon storage in grassland and forest soils, but led to a 3.5% increase in SOC in arable soils (Lu et al., 2011). Another study found that more moderate application than intensive nitrogen fertilisation increased SOC, as increased carbon inputs exceeded rates of carbon mineralisation (Soussana et al., 2004). A recent study in Northern Ireland found that significant changes in SOC stocks primarily occurred in the top 20 cm of soil, and only between the two extremes of nutrient treatment: “unfertilised” versus “highly fertilised” (Fornara et al., 2020).

There is evidence that reducing fertiliser use can yield a net benefit in terms of limiting GHG emissions, in some cases without adversely impacting SOC storage. (Soussana et al., 2007) suggest that in the absence of N supply and herbage use, grasslands are net carbon sinks. In one meta-analysis, fertilisation during or after land-use change to pasture had no significant impacts on SOC stocks (Guo & Gifford, 2002). Furthermore, fertiliser application may reduce the relative allocation of carbon belowground (Haynes & Gower, 1995). In an experiment at Broadbalk in Rothamsted, fertilisation was found to cause an increase in SOC, but the annual GHG emissions from nitrogen fertiliser were four times that of the carbon sequestered as SOC (Powlson et al., 2011). Thus, in there was no net climate change mitigation resulting from fertiliser use, although the increase in SOC would be beneficial for soil function. One study has suggested that a reduction in CO_2 emissions, associated with the energy-intensive and therefore fossil fuel-hungry nature of fertiliser manufacturing, could be achieved if the use of NPK fertilisers in grasslands were substituted for PK fertilisers and nitrogen fixation by legumes (Poeplau et al., 2018), although it should be noted that any reduction in productivity would drive displacement. A global meta-analysis of the effect of fertilisation on soil organic matter (SOM) found that the application of N fertilisers significantly slowed the rate of SOM loss in agricultural soils with cereal based crop rotations, but that the application of synthetic fertilisers alone did not prevent the loss of SOM completely (Ladha et al., 2011). The application of organic fertilisers was associated with a significant increase in SOM over time but does not necessarily represent genuine additional carbon sequestration.

3.29.3.2 Co-Benefits and Trade-offs

According to Bai et al. (2019), reduced nitrogen fertiliser inputs will likely lead to greater SOC storage benefits brought about by "climate-smart agriculture" interventions including:

- Use minimum-tillage or no-tillage cultivation [**ETPW-092**]
- Use green manures within the rotation [**ECCM-023**]
- Minimise bare soil to reduce soil loss e.g. cover crops, crop residues, trees coppice etc. [**ECPW-002**].

[TOCB Report 3-1 AQ **ETPW-242**] White clover living mulch plots have been shown to also have higher greenhouse gas fluxes.

[TOCB Report-3-3 Soils **ETPW-242**] Nutrient and manure management actions have implications for soil quality.

[TOCB Report-3-5A Croplands **ETPW-242**] The effects on biodiversity of **ETPW-242** are likely to be similar to those of **ETPW-252** and **ECCM-003**, both of which have full assessments under reduced fertiliser use (Report-3-5A Croplands). There are likely to be some benefits to biodiversity of reducing fertiliser use, but much of the published evidence is from studies reducing fertiliser, pesticide and herbicide use (Dicks et al. 2013), so the effects and magnitude of potential biodiversity benefits from reducing fertiliser use are not well understood.

RAG rating for specific ecosystem service (for **ETPW-242**):
AMBER L* maintaining species / wider biodiversity
AMBER TL* presence of rare and priority species

3.29.3.3 Magnitude

Freibauer et al. (2004) report sequestration of 0.2 t C ha^{-1} y^{-1} due to fertilisation of nutrient poor grasslands, but -0.9 – 1.1 t C ha^{-1} y^{-1} for intensification of organic soils. A literature review of the effects of grassland managements on SOC stocks, in conditions applicable to the UK found that the application of inorganic fertiliser resulted in carbon stock changes in grassland of between -21 to 27 t C ha^{-1} , across studies with variable (unstated) timeframes, for the top 30cm (approximately) of soil (Buckingham et al., 2013; Moxley et al., 2014).

3.29.3.4 Timescale

Unclear. Most of what we can say in terms of timescales is based on inferring from studies where fertiliser application increased. One study showed increases in SOC storage on newly created grassland that had manure and nitrogen fertiliser applied within three years (Amman et al., 2007).

3.29.3.5 Spatial Issues

A meta-analysis by Lu et al. (2011) found that nitrogen addition increased soil carbon in cropland, but not forests or grasslands. Reducing, or simply avoiding applying excess amounts of, fertiliser could be highly beneficial. Intensification of fertiliser use on organic soils was shown to produce SOC losses of between 0.9 and 1.1 t C ha^{-1} yr^{-1} (Freibauer et al., 2004). For mineral soils on the other hand, SOC tends to increase with application of fertiliser (Alison et al., 2019), so reducing fertiliser use here will likely require implementation of other interventions reviewed in the Soil Protection and Management bundle to succeed.

3.29.3.6 Displacement

Unclear (if done well) due to lack of sufficient evidence. If done poorly, the likely reductions in yield and productivity would increase pressures elsewhere.

3.29.3.7 Maintenance and Longevity

Unclear due to lack of sufficient evidence.

3.29.3.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.29.3.9 Climate Factors / Constraints

Lu et al. (2011) propose that under elevated CO_2, there may be increasing N limitation within ecosystems. The result might be that the positive effects of N fertilisation on SOC and carbon sequestration are increased under elevated CO_2.

N_2O emissions due to nitrogen fertilisation could be more severe in places where there is surplus nitrogen; N_2O emissions and yield-scaled N_2O emissions have both been shown to increase exponentially with nitrogen surplus for values between approximately 0 and 100 kgN ha^{-1} (although based on limited samples). Following this, failing to reduce nitrogen fertiliser application where there is a nitrogen surplus could mean disproportionately high N_2O emissions (van Groenigen et al., 2010). The official statistics for soil nutrient balances in England (DEFRA, 2021d) from report an average nitrogen surplus of 88.1 kg ha^{-1} in England in 2020.

3.29.3.10 Benefits and Trade-offs to Farmer/Land-manager

If done well, and the intervention results to the maintenance of productivity and yield, the farmer/land-manager would benefit from reduced costs associated with fertiliser application.

3.29.3.11 Uptake

If done poorly, and the intervention leads to reductions in productivity and yield, the farmer/land-manager will lose money and displacement would occur. Potentially, through incentives to pursue other interventions reviewed in the Soil Protection and Management bundle, the amount of fertiliser required may reduce, hence, farmers could be incentivised to reduce fertiliser use via these indirect means. According to Alison et al. (2019), contractors are increasingly employed to optimise fertiliser application using GPS, which could facilitate future reductions in fertiliser use overall.

3.29.3.12 Other Notes

None

3.29.4 **ECPW-109: Maintain optimum soil pH**

Duplicate evidence base: Apply lime only on neutral grasslands (lowland and upland hay meadows), with a soil test to maintain a pH of 6.0 [**ETPW-220**]

3.29.4.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**N**
	Below ground carbon sequestration	**LTD***

More evidence is needed in the UK context to conclude how effective maintaining optimum soil pH (especially via liming to increase pH on soils considered too acidic) is on soil carbon. As Alison et al. (2019)

argue, carefully targeted measures to maintain optimum soil pH can provide emissions reduction and increased productivity, while reducing manufactured fertiliser inputs.

It has been reported that in the UK, 40 % of all arable soils and 57 % of grassland soils are in urgent need for lime (calcium carbonate) application to raise their pH conditions from below the critical thresholds of 6.5 and 6.0 for the two land uses, respectively (Goulding, 2016). Optimising soil pH induced by soil acidification (which has had a recent legacy of acceleration due to human activities) is essential to enhancing nutrient availability, primary productivity, carbon and nitrogen mineralisation, and organic matter inputs to soil (Wang et al., 2021 and references therein). Research at Rothamsted, England reported that liming increased biological activity in the soil, and while soil respiration rates increased, these were offset by plant soil inputs being more readily incorporated into organo-mineral soil pools (Fornara et al., 2011).

Variable effects of liming on SOC stocks in grasslands have been noted, and liming has the co-benefit of increased productivity depending on soil type (Paradelo et al., 2015). Liming could increase the efficiency of NPK fertiliser effects on productivity, potentially reducing the need of such fertilisers and their negative externalities such as nutrient losses in runoff (Gibbons et al., 2014). Wang et al. (2021) found in their global meta-analysis that the changes in yield were largest in upland arable systems, at high liming rates and with longer term application (>6 years). Effects on yield were most significant where pH was <4.5 at the beginning (Wang et al., 2021).

When pH is raised via liming, there are associated CO_2 emissions (Gibbons et al., 2014). For every 2 mol of acid neutralised by calcium carbonate, up to 1 mol of CO_2 could be emitted (Whitmore et al., 2015). Liming is likely to be net carbon neutral due to the carbon footprint of lime mining, but may lead to increased crop productivity and reduced N2O and CH4 emissions overall (Wang et al., 2021). Another study proposed that CO2 emissions induced by lime application could be avoided by applying oxides (e.g. quicklime or slaked lime) instead of carbonates if they can be manufactured with CO2 recovery (Snyder et al., 2009). Silicates could also be used instead of carbonates for CO2 emission prevention (Whitmore et al., 2015).

3.29.4.2 Co-Benefits and Trade-offs

No assessment.

3.29.4.3 Magnitude

At Rothamsted, it was suggested that liming (4 t ha^{-1} every four years) can lead to SOC increases of 16.2 t C ha^{-1} over a period of roughly a century according to one study (Fornara et al., 2011). Furthermore, net SOC sequestration measured in the 0-23 cm layer was between 2 and 20 times greater in limed versus unlimed soils (Fornara et al., 2011).

The magnitude of the SOC storage effect will also depend on careful calibration of the optimum liming rate. Eze et al. (2018b) measured an insignificant increase in SOC stocks of 6.8 % and 2.8 % at low (<3 t ha^{-1}) and high (>5 t ha^{-1}) lime application rates. By contrast, SOC stocks increased significantly by 14.1 % under moderate (3-5 t ha^{-1}) lime application rates (Eze et al., 2018b).

3.29.4.4 Timescale

>10 years

The effects of liming on SOC could take a long time to materialise. At Rothamsted, beneficial impacts on SOC storage became detectable after approximately 50 years (Fornara et al., 2011). Monitoring under the UK Countryside Survey reveals that soil pH has become dramatically less acidic between 1978 and 2007 as sulphate deposition has declined, though there was no detectable change in topsoil organic carbon concentration (Thomas et al., 2020).

3.29.4.5 Spatial Issues

Care should be taken when generalising results of studies focussed on mineral soils to sites with organic soils. Experiments with short-term, high-dose liming on upland soils led to a decline in SOC, likely as a result of increased carbon decomposition outweighing plant carbon inputs (Rangel-Castro et al., 2004). Similar findings from upland grasslands in France were also uncovered (Lochon et al., 2018). As Alison et al. (2019) stresses, liming on organic soils may shift the system from low productivity and SOC decomposition under anaerobic conditions to one of high productivity and SOC decomposition rates under aerobic conditions – the effects of this on net CO_2 exchange being still unresolved.

3.29.4.6 Displacement

Unclear due to lack of sufficient evidence.

3.29.4.7 Maintenance and Longevity

Unclear due to lack of sufficient evidence, though it has been suggested that maintaining a minimum pH of 6 on mineral agricultural soils is important to maximise the benefits of fertiliser application (Chenu et al., 2019).

3.29.4.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.29.4.9 Climate Factors / Constraints

One review, covering field experiments in Europe, Asia, Australia, South America and North America, for forest, grassland and cropland soils found variable effects of lime additions, ranging from approximately – 20% of carbon under un-limed conditions to +70% of carbon under un-limed condition, combining data on both carbon stock and carbon concentration change. For sites which reported carbon stocks, liming effect varied between a stock change of –18.6 tC ha^{-1} and +17 tC ha^{-1} relative to the control, or emission (+)/sequestration (-) rates of 7.8 tC $ha^{-1}yr^{-1}$ to -0.5 tC $ha^{-1}yr^{-1}$. Overall, they observed no significant effect of liming on SOC, no significant effect of either duration of the experiment, lime application rate or soil texture on SOC storage, for all land uses (Paradelo et al., 2015). They concluded that the impacts of lime addition were context dependent, though there was insufficient evidence to predict the net effect on SOC accumulation under various soil, weather and land-use conditions (Paradelo et al., 2015).

3.29.4.10 Benefits and Trade-offs to Farmer/Land-manager

Improved productivity and nutrient use by plants, though contemporary evidence suggests these benefits are unlikely to occur on upland farms with organic soils.

3.29.4.11 Uptake

Investment costs continue to represent a barrier to maintaining optimum soil pH, though incentives have been offered in the past (ending in 1978) which increased the frequency of liming application compared to current rates (Gibbons et al., 2014; Holland et al. 2018). Future incentives for uptake could be very popular among farmers given the positive effects on productivity and nutrient use by plants (Alison et al., 2019).

3.29.4.12 Other Notes

More long-term studies of the effects of liming on SOC storage are required, especially in the UK context. Given the long timescales involved to detect SOC storage benefits, modelling approaches and further study of long-term monitoring data at co-located points (e.g. from the UK Countryside Survey) are also recommended. Indeed, liming effects are generally not included in biogeochemical and agroecosystems models, which hamper the applicability of these models for scenario studies to predict terrestrial ecosystem feedback to agricultural management changes and to identify innovative climate change-related mitigation and adaption strategies (Wang et al., 2021). Evidence gathering should also target upland soils where the opposite effects of liming on soil carbon have been noted on the relatively few studies we have to date.

3.30 Soil management and protection– Tillage

3.30.1 ETPW-092: Use minimum-tillage or no-tillage cultivation

Duplicate evidence base: Leave autumn seedbeds rough (instead of finely tilled seedbeds) [**EHAZ-018**]; Use direct drilling into crop stubble or cover crops [**ECPW-242**]

3.30.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**N**
	Below ground carbon sequestration	**LT***

Tillage of soil is a management practice typically applied to arable land to clear the accumulation of weeds, pests, prepare suitable conditions for seedling emergence and alleviate soil compaction (Gregg et al., 2021). Tillage has also been highlighted as a key driver of soil organic carbon (SOC) losses from agricultural soils, caused by disaggregation and aeration of the soil which accelerates SOC decomposition (Mikha & Rice, 2004). Thus, minimum-tillage and no-tillage cultivation have been recommended for carbon sequestration in soils (Alonso et al., 2012; Lal, 2004).

The effects of minimum-/no-tillage on carbon sequestration have been analysed and reviewed in several recent reports (Alison et al., 2019; Gregg et al., 2021; Maskell et al., 2019; Rollett & Williams, 2020). A meta-analysis by Bai et al. (2019) showed that the impact of conservation tillage practices, including minimum- and no-tillage, had a small but positive impact, increasing SOC by 5%. However, when bulk density and SOC distribution with depth are considered, the evidence for carbon sequestration in no-tillage systems is weaker (Angers & Eriksen-Hamel, 2008; Baker et al., 2007). In particular, adopting minimum-/no-tillage leads to changes in SOC content with depth, with higher levels at the soil surface (Alison et al., 2019; Powlson et al., 2014). Luo et al. (2010) noted that following the introduction of no-tillage, SOC increased by 3.15 ± 2.42 t ha^{-1} (mean ± 95% confidence interval) in the surface 10 cm of soil, but decreased by 3.30 ± 1.61 t ha^{-1} in the 20–40 cm soil layer, resulting in no overall improvement in SOC stocks for the top 40 cm. For these reasons, a review to inform LULUCF inventories stated that tillage reduction is not a reliable means of increasing SOC contents of UK soils (Moxley et al., 2014).

If tillage increases the distribution of SOC with depth, there will likely be positive impacts in terms of reduced SOC decomposition. Deeper mineral soil horizons typically contain lower amounts of SOC (Jobbágy & Jackson, 2000). Organic matter that is incorporated into deeper parts of the soil may degrade more slowly or become readily adsorbed onto fine mineral particles which will likely be less saturated than at the surface (Moxley et al., 2014). Consequently, deep ploughing to bury SOC-rich topsoil, as well as vertical redistribution of SOC by anecic earthworms have received interest (Chenu et al., 2019). However, there is a significant risk that the action of deep ploughing will cause increased SOC decomposition in topsoil, offsetting any stabilisation in deeper soil.

Ultimately, the roles of tillage in SOC reductions and adoption of minimum-/no-tillage practices need to be better understood. Reducing tillage intensity may lead to reduced soil bulk density with less disturbance and compaction (Gregg et al., 2021), although in the short term, it is generally observed that zero tillage causes increase bulk density (Soane et al., 2011). Monitoring SOC stock changes at fixed depths without considering any changes in bulk density has been shown to result in overestimating SOC in surface layers while underestimating at deeper layers (Xiao et al., 2020). It is for reasons such as these, along with claims that factors other than tillage have driven SOC losses on arable land – e.g. conversion to annual crops, periods of bare soil and drainage (Baker et al., 2007) - that have led some to conclude that the positive effect of no-tillage on SOC has been overstated, although increases in SOC from no-till in the long-term seem probable (Powlson et al., 2014; Stockman et al., 2013).

3.30.1.2 Co-Benefits and Trade-offs

No assessment

3.30.1.3 Magnitude

In the UK, the mean rates of sequestration arising from minimum-tillage are approximately -0.31 t C ha^{-1} yr^{-1} ± 0.18 t C ha^{-1} yr^{-1} (Powlson et al., 2012). This is comparable to a reported range of -0.3 to -0.4 t C ha^{-1} yr^{-1} for minimum-/no-tillage in Europe (Freibauer et al., 2004).

3.30.1.4 Timescale

Likely >10 years.
Potential positive impacts on carbon sequestration reported in the topsoil (up to 30 cm depth) over the course of 5 to 23 years (Powlson et al., 2012). Simulations using the SALUS model indicated that over a 15-year period, SOC losses were greater following conventional tillage than either minimum- or no-tillage (Cillis et al., 2018).

3.30.1.5 Spatial Issues

The effects of conservation tillage are also dependent on soil type and temperature (Alison et al., 2019; Luo et al., 2010) and there is also a risk of N_2O production in poorly aerated soils (Alison et al., 2019; Rochette, 2008).

3.30.1.6 Displacement

There is arguably a risk that if minimum-/no-tillage results in a drop in yield, more intensive farming will be required elsewhere to ensure food supply keeps up with demand (Alison et al., 2019).

3.30.1.7 Maintenance and Longevity

Increases in soil carbon storage may only occur if crop productivity can be maintained or increased (Snyder et al., 2009). A return to full tillage could reverse any positive effects of minimum-/no-tillage on soil carbon storage (Alison et al., 2019; Conant et al., 2007). Occasional tillage is sometimes applied in mainly zero till situations to control weeds; it is likely that this disturbance will negate all or some of any SOC accumulated during the period of zero till (Conant et al., 2007; VandenBygaart & Kay, 2004).

3.30.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.
Conservation tillage has been shown to increase SOC concentration by approximately 5%, with greater magnitude effects of no-tillage management in croplands situated in warmer climates (Bai et al., 2019). However, it has also been reported that the increases in SOC content and ultimately climate change mitigation via minimum-/no-tillage is small (Powlson et al., 2014). Indeed, such tillage practices are likely to primarily redistribute SOC in the soil, with some addition sequestration in the long term as mentioned above

(Powlson et al., 2014). Furthermore, there is a risk of increased N_2O emissions from poorly aerated soils which are common in the UK (Rochette, 2008).

3.30.1.9 Climate Factors / Constraints

The effects of conservation tillage, including on soil organic matter oxidation and seed germination, are dependent upon temperature and soil type (Luo et al., 2010). When no-tillage is combined with cover crops and crop rotation, its negative impacts on crop yields are minimised, especially in rain-fed dry climates (Pittelkow et al., 2015).

3.30.1.10 Benefits and Trade-offs to Farmer/Land-manager

Increased herbicide usage may be necessary, driving up costs for the farmer/land-manager, though there could be reduced fuel consumption costs too (Moxley et al., 2014). Possibilities of reduced yield or crop failure under minimum-tillage, with mixed findings for no-tillage on crop production (Freibauer et al., 2004). A review of the impact of no-till practices in northern Europe states that no-till usually allows earlier drilling of winter-sown crops but will give lower soil temperature and higher moisture content in spring, causing delayed drilling of spring-sown crops. It also requires that additional care be taken to minimise soil damage at harvest and to ensure the even distribution of crop residues prior to drilling (Soane et al. 2012).

3.30.1.11 Uptake

Recent analysis of tillage management in England revealed that 47.6% of arable land is cultivated using some form of reduced tillage without total inversion of soil and 7% under no- tillage, and that as farm size increased, the likelihood of adopting minimum-tillage increased (Alskaf et al., 2020). Alskaf et al. (2020) also list the following suggestions to increase uptake of minimum-/no-tillage practices in the future, including:

- Growing combinable crops (crops that are cut by a combine harvester, such as wheat, barley and oilseed rape). The majority of arable UK crops (by area) already satisfy this.
- Financial support to access minimum-tillage machinery.
- Support of a network of “farmer champions” to facilitate practical knowledge exchange

Adopting controlled traffic farming (CTF) can also facilitate the adoption of minimum-/no-tillage cultivation (Chamen, 2015).

3.30.1.12 Other Notes

Minimum-/no-tillage practices are intended to increase SOC, but current evidence is mixed, and uncertainty is high. SOC contents should be measured both in and below the plough layer, with bulk density monitored concomitantly to obtain more robust calculations of SOC stocks pre- and post-transition away from conventional tillage (Maskell et al., 2019).

3.31 Systems Action – Landscape Management

3.31.1 EBHE-233: Control scrub or trees to maintain views

Duplicated Evidence base: **EBHE-083** Remove and prevent the regrowth of scrub, bracken, sedge or reed and keep understorey vegetation trimmed back around scheduled monuments/ heritage assets on the shine database that are not Listed Buildings or Scheduled Monuments

Co-benefits and trade-offs only

3.31.1.2 Co-Benefits and Trade-offs

Food and fibre production	Area under production or yield and outside of ELM	N
Global, regional & local climate regulation	Above ground carbon sequestration	T**
	Below ground carbon sequestration	T*

There is good evidence that removing woody biomass can result in the loss of carbon from above and below ground biomass, due to soil erosion (Gregg et al., 2021; Matthews, 2020; Thomas et al., 2020). The magnitude of the negative effect on carbon sequestration will be dependent on the intensity of vegetation control, for example, whether plants are being removed or pruned. Axe et al. (2017) conducted one of the largest studies to date on hedgerow carbon stocks in the UK, and showed that hedgerows maintained in taller and wider states contained more carbon in above ground biomass, with a 4.2m wide hedge containing 9.7 tC km-1 more than a hedge 2.6m wide, with a mean height of 3.5m. Based on these findings Prosser et al. (2022) reports that increasing hedgerow height from 2.0 m to 2.7 m would represent an increase in size to 70 per cent of currently managed hedgerows across England and Wales, with a potential to sequester an additional 2.0 Mt carbon.

[TOCB Report-3-5D Systems **EBHE-083**] Some species will benefit from scrub being removed, but more will benefit from semi-natural scrub, so this will generally be detrimental to biodiversity. However, there has been no specific research on this and the scale of any effects will only be small.

3.32 Systems Action – Mixed Systems and Cross Habitat Action

3.32.1 ECCA-035: Prepare and implement wildfire management plans

3.32.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	N
Global, regional & local climate regulation	Above ground carbon sequestration	LTD**
	Below ground carbon sequestration	L*

Expert opinion and evidence for the potential impact of wildfires on carbon stocks in the UK, suggests that a wildfire management plan is likely to positively affect carbon sequestration potential. Estimates of emissions from forest wildfires from 2019 were estimated as 293.87 $GgCO_2$, although estimates were associated with low confidence (Brown et al., 2021; Element Energy & UKCEH, 2021). A wildfire management plan, ideally one informed by likely change in wildfire risk with climate change, would enable the risk of carbon loss to be minimised, and also for potential disturbance caused by the management itself to be appropriate to the given risk (Arnell et al., 2021). There has been little empirical study regarding the effectiveness of fuel breaks or vegetation management for the overall emissions balance for reducing the damage done by forest fires. A study by (Syphard et al., 2011) assessed whether fuel breaks successfully halted fires in California over a period of 26 years, and found a 46% success rate, that was strongly associated to the presence of additional fire suppression activities. Access by firefighters, fire size, larger fuel breaks, younger vegetation and good maintenance of fuel breaks were significantly associated with fire containment.

For assessments of the impact of fire in peat, heathland and woodland see actions 3.1.2.2, 3.11.5.0 and 3.11.9.5, respectively.

The forestry commission provides a template for a wildfire management plan, for which the stated objective are reducing the likelihood of a wildfire starting, reducing the severity of fire (i.e. the amount of organic matter burnt); minimising the burned area and the effects of fire on vegetation, and reducing impact on life, well-being, and assets of economic, cultural or ecological value, and finally to increase opportunities to suppress the fire (Building wildfire resilience into forest management planning - Forest Research).

3.32.1.2 Co-Benefits and Trade-offs

Not assessed

3.32.1.3 Magnitude

Sozanska-Stanton et al. (2016) estimate that stopping the prescribed burning of heathland and moor grass would reduce GHG emissions by 7 t CO_2eq ha^{-1} y^{-1}, using IPCC methods. Impacts due to wildfire risk are less certain.

The area affected by woodland fire in the UK each year is highly variable and the average area burned was 2139.6 ha yr^{-1} (423 to 8675 ha yr^{-1}) in the UK and 233.3 ha yr^{-1} (29 to 979 ha yr^{-1}) in England from 2010 to 2017[9]).

3.32.1.4 Timescale

>10 years
Greatest returns are likely to occur over long timescales, where there are risks of large fires.

3.32.1.5 Spatial Issues

Wildfire management plans could be broadly applied, although are likely to be more relevant in some circumstances than others. Administrative costs may make targeting this initiative appealing.

3.32.1.6 Displacement

Not assessed

3.32.1.7 Maintenance and Longevity

May require financial support to ensure planned management can be sustained long term and a sufficiently detailed plan be developed.

3.32.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.
Reducing wildfire risk could also mitigate the increasing likelihood of wildfire with climate change.

3.32.1.9 Climate Factors / Constraints

Not assessed

3.32.1.10 Benefits and Trade-offs to Farmer/Land-manager

Protection of assets and life. May incur opportunity costs due to land requires for wildfire management options (fire breaks etc.) as a result.

3.32.1.11 Uptake

Constructing a detailed wildfire management plan may require financial support and access to specialist expertise.

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[9]www.forestresearch.gov.uk/tools-and-resources/statistics/forestry-statistics/forestry-statistics-2018/environment/woodland-fires/

3.32.1.12 Other Notes

None

3.33 Systems Action – Pests and Disease Management

3.33.1 ETPW-265: Fell diseased trees where the action is uneconomic

3.33.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**LTD****
	Below ground carbon sequestration	**LTD***

There is a strong logic chain suggesting that the felling of diseased trees is likely to be net beneficial, if successful at preventing the spread of disease. However, there is a lack of evidence quantifying this, and the effectiveness of management is highly likely to vary with specific disease dynamics.

Tree disease can result in the loss of standing carbon stocks and sequestration potential due to death or morbidity in affected trees. If not managed, mortality and morbidity from tree disease can be extensive (Quirion et al., 2021). Subsequent regeneration at the site may be hampered follow tree felling if a source of pests or disease remains (Demeter et al., 2021; Turczański et al., 2021.)[10] Recent research in the US has also shown that disease and pests can also cause a significant reduction in carbon sequestration in living trees (Quirion et al., 2021) estimating a 28% smaller change in above ground biomass in live trees where a disease has affected at least 0.4ha, compared to trees where no disturbance was recorded affecting at least 0.4 ha in similar ecological contexts and with similar starting above ground biomass prior to disturbance, based on a national forest inventory plot data of US forests. Ash die back has been estimated to have cost Britain approximately £15 billion in management costs and lost ecosystem services (Hill et al., 2019).

Despite good evidence for a potentially large effect of tree disease on carbon sequestration potential, there is mixed evidence for the effectiveness of felling diseased trees in presenting disease spread. Research into the best management of Ash die back is still associated with a high degree of uncertainty. Thinning had more positive consequences than felling affected areas and on subsequent regeneration in the short term, but in the long term (50-100yrs) there was little difference, although confidence in these findings was deemed low (Mitchell et al., 2014). The felling of trees affected by acute oak decline is recommended (Denman et al., 2010).

For felling diseased trees to be effective it should be paired with good monitoring so signs of disease can be identified early. Ideal management is likely to vary with disease, particularly when new emergent diseases enter the UK. The felling of diseased trees can be mandated using a SPNH[11].

3.33.1.2 Co-Benefits and Trade-offs

No assessment.

3.33.1.3 Magnitude

[10] www.gov.uk/guidance/replace-trees-after-felling-due-to-pests-and-diseases
[11] www.gov.uk/guidance/replace-trees-after-felling-due-to-pests-and-diseases

Ash die back has been estimated to have cost Britain approximately £15 billion in management costs and lost ecosystem services (Hill et al., 2019). Recent research in the US has also shown that disease and pests can also cause a significant reduction in carbon sequestration in living trees (Quirion et al., 2021) estimating a 28% smaller change in above ground biomass in live trees where a disease has affected at least 0.4ha, compared to trees where no disturbance was recorded affecting at least 0.4 ha in similar ecological contexts and with similar starting above ground biomass prior to disturbance, based on a national forest inventory plot data of US forests.

However, the effectiveness of felling in controlling the effect of disease spread is likely to be highly variable and has not been quantified in terms of the impact on carbon sequestration.

3.33.1.4 Timescale

Variable. There will be short term effects due to the direct effects of felling activity including the loss of living biomass and soil carbon stocks due to disturbance. In the medium to long term, there may be an increase in carbon sequestration rates due to a reduction in the size of the disease pool or probability of transmission, however this effect may be difficult to quantify.

3.33.1.5 Spatial Issues

The spatial and temporal window in which felling diseased trees is effective management is likely to be variable across diseases and require informed targeting.

3.33.1.6 Displacement

No assessment.

3.33.1.7 Maintenance and Longevity

Any felling of diseased trees should be supported by ongoing monitoring efforts of disease presence.

3.33.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

3.33.1.9 Climate Factors / Constraints

No assessment.

3.33.1.10 Benefits and Trade-offs to Farmer/Land-manager

No assessment.

3.33.1.11 Uptake

Various licences are required to fell disease trees, [Fell diseased trees - GOV.UK (www.gov.uk)] and a lot of regulations are involved, and will need to be appropriately navigated.

3.33.1.12 Other Notes

None

4 Key Action Gaps & Other Evidence

4.6 Actions with Assessment of Trade-off/Co-benefits only

4.6.1 ETPW-043: Restrict beach cleaning to hand removal of inorganic waste

4.6.1.2 Co-Benefits and Trade-offs

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**N**
	Below ground carbon sequestration	**N**

There is no evidence to suggest that organic matter on beaches contributes to carbon sequestration. Whilst the carbon sequestration potential of macro algae as an important blue carbon sink (Macreadie et al., 2019), beaches are highly unlikely to retain any carbon stocks long term.

4.6.2 EN17033 [ECPW-281] Use of biodegradable silage, crop cover mulches and planting trays to meet recognised compostable standard

Duplicated evidence base: Use alternatives to fossil-based plastic mulches, such as green mulches or other biodegradable materials. Straw, shredded wood and other natural products can also be used as mulch [**ECPW-280**]

4.6.2.2 Co-Benefits and Trade-offs

The impacts of biodegradable plastic substitutes, including mulches, on carbon cycling are not yet well understood. The substitution of traditional fossil fuel derived products with biomass-derived products would constitute emissions abatement. However, compostable standard EN17033 requires the ≥90% conversion of mulch's carbon into CO2 within 2 years under ambient soil conditions, implying that a significant enhancement stocks from the addition of carbon rich material in this way is unlikely (Douglas G. Hayes & Markus Flury, 2018). Traditional mulches have been associated with lower rates of soil carbon accumulation than controls, whereas one study of biodegradable mulches showed no effect on rates of soil carbon accumulation relative to control (English, 2019). The degradation of C rich materials has the potential to enrich soil carbon stock, or may have a priming effect on rates of soil respiration (Sayer et al., 2011). Analysis by English (2019) found the carbon content supplied by the degradation of C rich mulches was marginal compared to crop residues. The effect of soil inputs on the microbiome and microenvironment of soil may have consequences for carbon cycling in the long term, but estimates of effect size or direction are lacking (Serrano-Ruiz et al., 2021).

Food and fibre production	Area under production or yield and outside of ELM	N
Global, regional & local climate regulation	Above ground carbon sequestration	N
	Below ground carbon sequestration	LD*

[TOCB Report-3-5D Systems **ECPW-280**] There have been no direct tests of effects of plastic mulch removal on biodiversity, but one study in Poland showed that mulching with plastic foil had negative effects on the number of species and abundance of farmland birds at local and landscape scales (Skorka et al. 2013). Mulching with plastic foil had a negative effect on potential resources for birds including adult butterflies and their larvae and weed species.

4.7 New Actions

4.7.1 New Action "Carbon_01": Conservation of long-established woodlands with existing high carbon stocks

4.7.1.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	
Global, regional & local climate regulation	Above ground carbon sequestration	***
	Below ground carbon sequestration	**

There is very strong evidence that preserving existing carbon stocks in woodland will result in a large emissions abatement, however not all existing woodlands are expected to experience a substantial risk of loss. Therefore, targeting conservation action where the risk of loss is highest will provide the greatest benefit.

The carbon sequestration rates of contemporary woodland are largest of any semi-natural habitat (Gregg et al., 2021). In a review of carbon dynamics across semi-natural and managed habitats in the UK, Gregg et al. (2021) estimate average soil and vegetation carbon stocks that could be achieved for mixed, native broadleaved woodland at 30 and 100 years of age (see Table 1). Estimates of contemporary carbon stocks throughout woodland and across separate countries in the UK were made by Matthews (in prep) and are reproduced in Table 2. Potential causes of woodland loss include deforestation, disease, pests, and fire.

The permanent conversion of forest systems to non-forest systems (particularly agriculture) is associated with the loss of a large proportion of sequestered carbon, and an increase in in situ emissions due to subsequent agricultural activity. The clearfelling of a forest without subsequent regeneration represents and extreme end of the spectrum of thinning intensification, which as discussed in section 3.24.2, result in a decrease in carbon stocks and accrual of carbon debt, in addition to emissions associated with felling (Matthews in prep). In situations where the conversion of forest to non-forest land uses is likely, retaining woodland would have a large benefit for net carbon sequestration. In an analysis of woodland GHG mitigation potentials in Wales, (Matthews, 2020) estimated that the avoidance of woodland loss would constitute the mitigation of -121 tCO_2-eq ha^{-1} yr^{-1} for woodland reserve and -56 tCO_2-eq ha^{-1} yr^{-1} for production woodland. The size of this benefit is constant over all time horizons, assuming the extent and condition of existing woodland is stable. Preventing the conversion of existing woodland with large carbon stocks to production woodland was estimated to amount to the sequestration of -5.5 tCO_2-eq ha^{-1} yr^{-1} on average, by 2050. However, current rates of permanent woodland area loss in the UK are relatively low.

4.7.1.2 Co-Benefits and Trade-offs

Not assessed

4.7.1.3 Magnitude

In an analysis of woodland GHG mitigation potentials in Wales, (Matthews, 2020) estimated that the avoidance of woodland loss would constitute the mitigation of -121 tCO_2-eq ha^{-1} yr^{-1} for woodland reserve and -56 tCO_2-eq ha^{-1} yr^{-1} for production woodland. The size of this benefit is constant over all time horizons, assuming the extent and condition of existing woodland is stable.
However, the opportunity for this initiative will be limited for circumstances where there are significant risks of deforestation in the absence of such mitigation measures. Current rates of permanent woodland area loss in the UK are relatively low.

4.7.1.4 Timescale

<5 years
Impact of preventing deforestation and woodland clearance wood be immediate.

4.7.1.5 Spatial Issues

Broadly applicable
Pressures on woodland are likely to be spatially variable.

4.7.1.6 Displacement

If woodland is effectively conserved but the drivers of deforestation remain then it is likely displacement will occur. If demand is for wood products then displacement would occur outside of the scheme. However, if the driver was conversion to a second land cover type this may occur on non-woodland habitat within the scheme.

4.7.1.7 Maintenance and Longevity

The continued conservation of existing woodland, particularly old growth woodland, is required to maintain carbon stocks.

4.7.1.8 Climate Adaptation or Mitigation

Any net carbon sequestration constitutes climate change mitigation.

4.7.1.9 Climate Factors / Constraints

Not assessed

4.7.1.10 Benefits and Trade-offs to Farmer/Land-manager

Landowners will experience continued benefits from woodland ecosystem services.
However, there may be opportunity costs from not pursuing alternative land uses and sale of wood products.

4.7.1.11 Uptake

Not assessed

4.7.1.12 Other Notes

Not assessed

4.7.2 New Action “Carbon_02”: Longer rotations in even-aged managed stands*

4.7.2.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	*
Global, regional & local climate regulation	Above ground carbon sequestration	TD**
	Below ground carbon sequestration	*

The extension of woodland rotations beyond the age of maximum mean annual increment was proposed by Matthews (2020) and Matthews et al. (in prep.), following analyses of the GHG mitigation potential of woodland in Wales and England and Scotland, respectively. For Wales, Matthews 2020 estimated that the extension of rotation length would provide the additional net sequestration of -0.8 tCO_2 eq ha^{-1} yr^{-1} by 2050. Matthews et al. (in prep.) modelled the effect of increasing the rotation length of yield 12 class Sitka spruce from 54 years to 74 years. This was associated with a 6% reduction in net wood carbon increment, from 3.4 tC ha^{-1} to 3.2 tC ha^{-1} due to slower rates of carbon accumulation past the maximum mean annual increment, which sets traditional rotation lengths in most commercial forestry. However, mean carbon stocks increase by 46% when rotations are extended, from 70tC ha^{-1} to 102 tC ha^{-1}, due to the long term average age of trees present at the site increasing and when averaging over multiple forestry sites at different points in rotation (Matthews et al., in prep.). Whilst rates of carbon production reach a maximum at MAI, carbon stocks continue to increase over much longer time scales (Matthews et al., in prep.). The range of carbon stocks that occur over the range of ages at which the annual increment is near the MAI, is shown for fast and slow growing varieties of sitka spruce and oak in Table 5, reproduced from Matthews et al.(in prep.). As pointed out by the authors, carbon stocks at the highest age in proximity to MAI can be 200% larger than those at the lowest age. Slower growing species typically have much longer windows of time where productivity is approximately that of the MAI (70 to 80 years in slow growing oak woodland), whereas the window is much narrower for fast-growing species (35 years for yield class 24 Sitka spruce) (Matthews et al., in prep.), therefore the potential benefits and consequences for wood production are yield specific.

We are not aware of any practical applications of this management in the literature. Although, the evidence base underpinning this modelling work is considered to be high quality, practical testing of this management before widespread application is desirable. Further consideration may also be warranted for the impacts of extending rotations of wood supplies in the short and long term. There is also a greater loss of losing biomass as a result of wind damage as age increases (Rich et al., 2007).

4.7.2.2 Co-Benefits and Trade-offs

There will be some trade-offs with potential for timber and biomass supply from forests. This may present some economic impacts for the forest sector and will restrict the availability of timber and biomass for substituting for more GHG-intensive materials and fuels.

4.7.2.3 Magnitude

The effects of increasing harvest age from the minimum to the maximum age at which productivity approximates MAI is shown by the mean carbon stock ranges given in Table 5, as estimated by Matthews et al., in prep..

Table 5 Table reproduced from Matthews et al. (in prep.). Examples of ranges of mean carbon stock estimates over a range of rotation ages consistent with near-maximal mean net woodland carbon increment.

Tree species	Yield class	Management	Age range (years)	Mean carbon stock range (tC ha^{-1})
Sitka spruce	24	No thinning	30-65	63-189
		With thinning	35-70	56-116
	12	No thinning	40-80	39-119
		With thinning	45-85	33-72
Oak	6	No thinning	45-115	34-123
		With thinning	50-130	31-80
	4	No thinning	55-125	28-90
		With thinning	60-135	26-60

4.7.2.4 Timescale

The impact of extending rotation length on carbon stocks will accumulate for each year beyond the original rotation date, but the largest benefits will on decadal timescales.

4.7.2.5 Spatial Issues

Not assessed

4.7.2.6 Displacement

Interactions between delaying harvests and demand for wood products could be complex, depending on the scale and timing of implementation, and could potentially displace wood production nationally or internationally, or cause substitution for non-wood products with associated implications for net GHG balance.

4.7.2.7 Maintenance and Longevity

It will be necessary to adhere to longer rotation lengths in perpetuity for increases in mean carbon stocks to be maintained. Any subsequent shortening of rotation length will erode carbon stocks to a lower average level.

4.7.2.8 Climate Adaptation or Mitigation

This activity supports climate change mitigation.

4.7.2.9 Climate Factors / Constraints

Not assessed

4.7.2.10 Benefits and Trade-offs to Farmer/Land-manager

Delaying harvesting will increase the total volume of wood collected at harvest, but will fractionally decrease the overall efficiency of the harvest.

Delaying harvest dates relative to original expectation may cause financial challenges for land owners /managers in the interim, and may increase the risk of losing stands to other causes.

4.7.2.11 Uptake

Not assessed

4.7.2.12 Other Notes

None

4.7.3 New action "Carbon_03": Create and implement woodland carbon plan

4.7.3.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	
Global, regional & local climate regulation	Above ground carbon sequestration	T
	Below ground carbon sequestration	T

The climate change mitigation benefits of woodland creation and management (including existing woodlands) are most effectively realised if implemented holistically and matched to local conditions (Matthews, 2020; Matthews et al., in prep.).
The idea of a woodland carbon plan is to incentivise well thought-through coordinated actions to create new woodlands, and to manage existing woodlands. Activities aimed at improving the management of existing woodlands can be very diverse, reflecting the diversity of possible definitions of 'improved' or 'sustainable' woodland management, and consequent management objectives, which may have nothing to do with carbon stocks. Depending on the objectives, appropriate management may involve increasing, maintaining or reducing the growing stock (and therefore carbon stocks) in woodlands. Hence, general commitments to conform to 'good practice', or 'improved' or 'sustainable' forest management (such as covered by UK Forestry Standard) do not necessarily (certainly do not automatically) lead to a conservation or enhancement of woodland carbon stocks in all situations. As a result, we suggest the development of a woodland carbon plan to explicitly address this need.

The objective of a woodland carbon plan would be to facilitate the achievement and/or maintenance of defined levels of carbon stocks on average across a collection of woodlands or woody vegetation, i.e. at a farm, landscape or regional scale. Whilst carbon stocks in individual woodlands could vary considerably (e.g. depending on the particular ages of stands of trees or managements needs such as when woodlands are thinned or clearfelled), landscape average carbon stocks can been maintained more consistently at a defined

target level if activities coordinated. The approach provides a mechanism for managing long term reductions in the local carbon stocks of woodlands where management objectives require, for example, to increase the harvesting of biomass and timber or to create a range of habitats within woodland areas, whilst maintaining landscape sequestration by offsetting this carbon loss elsewhere.
The development of a woodland carbon plan encourages farmers and landowners to think at a broader scale about how a coordinated approach to the management of existing woodlands and creating new woodlands could contribute towards climate change mitigation, by both sequestering carbon directly in woodlands and by increasing available renewable timber and biomass resources, whilst avoiding trade-offs that can occur when trying to achieve these objectives (Matthews, 2020; Matthews et al., in prep.).

The objectives are thus to:

- Support the creation of new areas of woodland to enhance carbon stocks and, potentially, increase the available biomass and timber resource
- Ensure that carbon stocks in existing woodlands are not compromised by long-term and progressive depletion, but are maintained, and where appropriate, are enhanced
- Support the management of woodlands to mobilise timber and woody biomass resources to contribute towards mitigation in other sectors such as construction, furniture and energy (bioenergy) whilst avoiding negative impacts on overall woodland carbon stocks.

This would be achieved through a coherent coordinated plan involving a tailored set of activities, including those specified elsewhere in this assessment, such as woodland creation and the enrichment of woodland growing stock.

A full discussion of the approaches that might be adopted is out of scope but, in summary, an example approach might involve the following steps:

- Defining the areas of woodland and non-wooded land covered by a specific woodland management plan, for example this might consist of the land owned by a particular individual or company.
- Stratifying the woodland areas into stands of regular species composition, growth rate and management prescription being applied (if any).
- For each stand, estimating the 'effective' long term average carbon stock per hectare associated with the stand, allowing for the management prescription actually being prescribed. (The management prescription would need to be obtained from historical data and/or a pre-existing management plan for the stand.)
- For each stand, estimating a 'benchmark' average carbon stock per hectare associated with the stand, assuming the stand was managed according to a standard, 'benchmark' prescription. (In some situations this might be the same as the management prescription actually applied to the stand.)
- Specifying a management plan which gives full details about how the management prescriptions (if any) applied to different stands are going to be changed.
- For each stand, where appropriate, estimating a 'modified' long term average carbon stock per hectare associated with the stand, assuming the management of the stand was changed according to the management plan. (In some situations this might be the same as the management prescription actually applied to the stand.)
- Defining areas of non-wooded land intended for conversion to woodland, and specifying how these woodlands are to be managed. For each new woodland, estimating a long term average carbon stock per hectare associated with woodland, assuming the management of the stand follows the management plan.
- Combining the per-hectare long term carbon stock estimates derived above with the areas of each stand/woodland, then adding them together to obtain estimates of total carbon stocks for the complete collection of woodlands, based on the existing situation and the planned changes, to demonstrate how woodland carbon stocks will be conserved or enhanced.

- Integrating the assessment of carbon stocks into a consideration of woodland mitigation that takes account of cross-sectoral mitigation effects (i.e. provision of woody biomass for use in construction and for bioenergy).

The approach proposed here is not without precedent in international literature, with related ideas being put forward in the past by Kirschbaum et al. (2001), Maclaren (2000) and Robertson et al. (2004). However, it should be noted that the integration of assessment of carbon stock impacts alongside cross-sectoral mitigation impacts has not been considered although, in principle, the methods could be extended to cover this. More recently the concept of 'climate smart forestry' has been proposed and demonstrated in some regions (Nabuurs et al., 2018; Bowditch et al., 2020; Verkerk et al., 2020); the notion of supporting the creation of woodland carbon plans is consistent with the principles of climate smart forestry.
There would need to be a way of ensuring that there are no perverse incentives arising from introducing such a scheme, e.g. for landowners to reduce carbon stocks in woodlands before they enter into a scheme for conserving carbon stocks. (This might be achieved by stipulating that historical as well as current data on stand management is provided as part of the classification of stands according to the management prescription.)

A reliable and independent system of verification would be needed to support effective implementation.
Methodologies, supporting tools and underpinning data would be needed to support both the development and implementation of woodland carbon plans and also to underpin verification methodologies.
Provision may need to be made to address potential risks of leakage due to some land areas being subject to control of forest carbon while others may not be. Several options exist for addressing this issue, the most extreme being to make the implementation of the instrument and measure mandatory. Alternatively this could be supported through continuation of existing regulation of tree felling, including as part of thinning operations, by ensuring that licenses are obtained for such activities. Checks could be made on risks of leakage as part of the issuing of licenses.

4.7.3.2 Co-Benefits and Trade-offs

Any co-benefits are likely to be specific to sites and contexts, but often there can be wider benefits through enhancement of landscape diversity and/or site/soil remediation or protection. The activity should have wider value through the protection of the biodiversity of established woodlands (e.g. through creation of a greater variety of woodland types). There may also be cross-sectoral benefits if the new woodlands are managed for biomass and timber production, displacing fossil fuels and emissions-intensive materials in the energy and construction sectors. Care may be needed to avoid woodland management or creation activities taking place on sensitive sites, or to ensure consistency of management with wider objectives (e.g. on sites of special scientific interest). Generally, some co-benefits should be supported and trade-offs avoided by ensuring that woodland management and creation is consistent with the existing UK Forestry Standard.

4.7.3.3 Magnitude

Matthews et al. (in prep.) report model estimates for 11 contrasting options for woodland creation involving tree species mixtures and management practices relevant for the UK. These options include natural recolonisation of non-wooded land with broadleaf trees and 'light' subsequent management, examples of mainly moderate- and fast-growing commercial coniferous woodlands, and managed woodlands composed of complex tree species mixtures. The mitigation contributed by the woodland options contrasts in magnitude over time, with different options providing the most mitigation benefits at different times (between 2022 and 2100 and beyond) and in different ways (involving direct carbon sequestration or GHG emissions savings through provision of wood products, to varying extents). It should be noted that the different woodland options are not always interchangeable on the same site or in the same location within the UK. Planting 1 hectare of woodland in 2022 is estimated to result in net carbon sequestration in woodlands and wood products over the period from 2022 to 2050 of between -0.8 and -13.8 $tCO_2\ ha^{-1}\ y^{-1}$, with a mean estimate for all 11 options of -5.7 $tCO_2\ ha^{-1}\ y^{-1}$. If GHG emissions savings arising from utilisation

of additional wood products and wood fuel are allowed for, these estimates increase to between -1.7 to -32.0 tCO_2 ha^{-1} y^{-1}, with a mean estimate for the 11 options of -12.4 tCO_2 ha^{-1} y^{-1}.
Systematic assessments explored in Matthews et al. (in prep) indicate that a package of woodland options matched to local site and climatic conditions can contribute carbon sequestration over at least 100 years of between -0.7 to -3.6 tC ha^{-1} yr^{-1} (-2.7 to -13.2 tCO_2 ha^{-1} yr^{-1}), depending on the local conditions and the shorter-term period considered within the 100-year time horizon. If carbon sequestration in wood products is included, these estimates are increased to between -1.2 and -3.9 tC ha^{-1} yr^{-1} (-4.2 and -14.3 tCO_2 ha^{-1} yr^{-1}). Also allowing for wood product displacement effects and wood cascading gives high estimates of the total GHG emission mitigation of between -1.7 and -6.0 tC ha^{-1} yr^{-1} (-6.3 and -22.0 tCO_2 ha^{-1} yr^{-1}).
In terms of specific management activities forming components of a woodland carbon plan, a quantitative assessment based principally on findings of Morison et al. (2012), Matthews (2020) and Matthews et al. (in prep) gives the following broad per-hectare estimates for the climate change mitigation potentials of woodland management activities:

- Estimates for woodland creation activities are given above. The actual mitigation realised on given site depends on the local conditions and the period considered within a 100-year time horizon. If woodland carbon “reserves” are created, this mitigation comes entirely from woodland carbon sequestration; if woodlands are created for wood production, then a proportion of this potential is contributed by carbon sequestration in products and product substitution.
- Adjustments to the management of existing woodlands to conserve or enhance woodland carbon stocks and sequestration can mitigate between -1 and -2.5 tCO2-eq. ha−1 yr−1 over a 100-year time horizon.
- Adjustments to the management of existing woodlands to increase wood production can mitigate between 0 and -3 tCO2-eq. ha−1 yr−1 over a 100-year time horizon through product substitution. Generally, this is more than offset by increased emissions (or reduced carbon sequestration) in woodlands of between -0.5 and -5 tCO2-eq. ha−1 yr−1 over the 100-year period, because of the impacts of increased harvesting in woodlands. However, the aim would be to compensate for this in other elements of the woodland carbon plan.
- Adjustments to the species composition and growth rates of existing woodlands, to enhance wood production whilst maintaining carbon stocks, give variable outcomes. Relatively high climate change mitigation potentials are estimated for activities involving the introduction of tree species or varieties with superior growth rates (e.g. genetically improved Sitka spruce), at -3 tCO_2-eq. ha^{-1} yr^{-1} over the period 2020 to 2050 and -5.5 tCO_2-eq. ha^{-1} yr^{-1} over the period 2020 to 2100 (the latter estimate including a significant contribution from product substitution).

The share of carbon sequestration benefits contributed above ground and below ground is variable and depends on the sites involved and the detailed woodland types and their management. It is likely to vary between a 50:50 share and an 80:20 share (above-ground:below-ground). The soil types involved in woodland creation and management activities, and the management of those soils prior to implementation of the woodland carbon plan, will largely determine these shares.

4.7.3.4 Timescale

Based on the assessments in Morison et al. (2012), Matthews (2020) and Matthews et al. (in prep), it is estimated that:

- Changes to the management of existing forests should be relatively easy to implement and carbon sequestration benefits often start to be achieved within 5 years and can continue over several decades. This can be important because existing forests contain significant existing carbon stocks.
- The main carbon sequestration benefits of woodland creation activities are achieved after 10 years but can continue for up to 100 years or longer.

- Climate change mitigation benefits arising from increased supply of biomass for use in construction and for bioenergy, substituting for more energy-intensive materials and fuels can be achieved quickly and can be long lasting, but the specifics are highly context specific.

4.7.3.5 Spatial Issues

In principle, a woodland carbon plan could be developed for any area of land or a defined collection of land areas (e.g. those owned or managed by an individual or a business). The effectiveness of the action is likely to increase if implemented at larger scales, because this would provide more flexibility for implementing different types of woodland-based mitigation activities in different areas, which together would contribute towards achieving the overall goals of improving woodlands, sequestering carbon and enhancing sustainable supplies of timber and biomass.

The relevance of specific activities within an individual woodland carbon plan (e.g. woodland creation or specific activities supporting the enrichment of woodland growing stock) will be very specific to localities and contexts. Relevant local factors include the extent of the existing woodland area within a locality, the tree species composition of the existing woodlands, extent to which those woodlands are already managed, and the suitability of land under other uses for woodland creation.

4.7.3.6 Displacement

Agricultural production could be lost at national scale if woodland creation activities form a significant part of woodland carbon plans. Changes to the management of existing woodland are likely to have marginal impacts on agricultural production, nationally or internationally, as long as levels of wood production from woodlands are at least maintained at or close to existing levels.

4.7.3.7 Maintenance and Longevity

Enhanced carbon stocks in woodlands are potentially impermanent.
Support may need to be given to landowners and their agents undertaking woodland carbon management activities when their forest areas are affected by disturbance events outside of their control.

If circumstances require woodland carbon plans to be modified or abandoned, then a mechanism is needed to ensure that compensatory actions are taken to conserve carbon sequestration and carbon stocks at target levels at the landscape/regional scale. These actions may involve the landowners involved or other agents.
Climate change mitigation achieved through enhanced supply of woody biomass for construction and bioenergy, potentially substituting for more GHG-intensive materials and fuels, should be permanent.

4.7.3.8 Climate Adaptation or Mitigation

A woodland carbon plan is directly aimed at supporting climate change mitigation but has co-benefits for adaptation.

Such a plan should also support the resilience of woodland and the wider landscape by diversifying land use and spreading risk. The plan could include provisions to show how this is addressed.

4.7.3.9 Climate Factors / Constraints

There are risks that the climate change mitigation benefits and co-benefits provided by managing woodlands with existing tree species (particularly where these are monocultures) and by creating new woodlands with certain tree species could be undermined if the climate becomes unsuitable for the tree species in the future. The woodland carbon plan could include provisions to show that risks arising from future climate change to the effectiveness of the planned actions has been assessed and that mitigation measures are identified (where needed).

4.7.3.10 Benefits and Trade-offs to Farmer/Land-manager

Farmers and land managers can potentially benefit from a more diversified source of income on their land and from resultant spread of business risk. However, creating new woodlands on land can also limit future options because usually there is a requirement to maintain woodland areas indefinitely, once they have been created.

4.7.3.11 Uptake

Culturally, farmers can be uncomfortable with woodlands, which are not recognised as an agricultural activity and are not identified as a potentially viable source of income (Lawrence and Dandy, 2014). New woodlands take farmland out of production and the irreversibility of woodland creation (see immediately above) is perceived as limiting options for land management and increasing business risk.

Landowners and farmers may perceive a risk being associated with joining together with others to develop larger scale woodland carbon plans, because they will be dependent on other participants to fulfil their contributions to the targets specified in the plan. This may be viewed as presenting a liability to achieving the overall goals of the plan which is outside the control of individual participants.

'Buy in' would be needed from a range of key stakeholders, e.g. environmental groups who may be concerned about risks of perverse incentives for land management, whilst industry groups may be concerned about excessive constraints on woodland management.

A number of options are available to address this issue including direct communication, stakeholder participation in the development of the detailed methodologies and demonstration of good practice in a network of exemplar projects.

4.7.3.12 Other Notes

The participation by owners of small areas of land may need to be facilitated. One way of achieving this might be by providing a central scheme which can be joined by such owners and operated on a collective basis.

4.7.4 New Action "Carbon_04": Enrichment of woodland growing stock

4.7.4.1 Causality

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**
	Below ground carbon sequestration	**TD***

The enrichment the growing stock of woodlands covers a set of activities that aim to enhance the standing biomass and carbon stocks, growth potential of existing woodlands, and to support their resilience to potential threats such as from storms, pests and diseases. Enrichment activities can be categorised very broadly into the classes of 'woodland restoration', 'woodland diversification' and 'enhancement of woodland productivity'.

Woodland restoration is "the process of reversing the degradation of … [woodlands] … thereby regaining their ecological functionality" (Besseau et al. 2018). Woodland restoration can involve increasing the number and variety of tree species in wooded areas of gardens, farms, fields, and forests, or facilitating natural regeneration of overgrazed, polluted or otherwise overused woodlands. Stanturf et al. (2014) provide a detailed review of forest restoration measures. Restoring forests can offer many benefits, including climate mitigation and adaptation, biodiversity conservation, support for other ecosystem services, local livelihoods, and well-being as well as economic gains. Challenges to forest restoration include a shortage of tree stock for planting, lack of practical guidelines and tools, including on site species matching, and aligning restoration activities with the needs and expectations of affected communities.

Woodland diversification aims to increase the complexity of woodland areas. Woodland management options include conversion from even-aged stands and clearfelling to “continuous-cover forestry” (CCF), i.e. continuous thinning to develop a more complex, two- or multi-storey canopy structure. Diversifying tree species in woodland and the development of more diverse mixtures of tree species in woodlands are identified as presenting opportunities to increase the potential for biomass growth and production in woodlands (Hulvey et al., 2013; Jactel et al. 2018; Kelty, 2006; Mason and Connelly, 2018; Reynolds et al., 2021).

The enhancement of woodland productivity can be achieved through the activities described above and also by introducing more productive tree species in existing woodlands (alongside existing tree species where appropriate), including through better matching of tree species to sites and climate and improvements to silviculture (Lee and Matthews, 2004; Mochan et al., 2008).

The above evidence suggests that woodland enrichment activities can provide significant benefits in terms of carbon sequestration, potential biomass supply from woodlands and the resilience of woodlands to environmental threats. However, these activities require expert knowledge of the potential of different tree species to grow on different sites, singly and in mixtures, and careful planning as part of implementation (Ennos et al, 2020; Kerr et al., 2020; Reynolds et al., 2021).

It is important to note that the above activities are not mutually exclusive – it is possible that in some situations activities may involve a synergistic combination of woodland restoration and diversification which also provides enhanced woodland productivity. However, these overlaps may make it difficult to classify exactly what actions are being taken and what benefits are being delivered, so that the appropriate level incentives for these activities can be determined.

There may be politically challenging issues in relation to supporting some activities. A key case is providing incentives to restore degraded woodlands, where landowners or managers may have previously been responsible for allowing woodlands to get into a degraded state (even if perhaps unconsciously).

4.7.4.2 Co-Benefits and Trade-offs

Any co-benefits are likely to be specific to sites and contexts, but often there can be wider benefits through enhancement of woodland quality and diversity and/or site/soil remediation or protection. The activity should have wider value through the protection of the biodiversity of established woodlands (if woodland restoration and diversification activities are carried out appropriately). There should also be cross-sectoral benefits where woodlands are managed for biomass and timber production, displacing fossil fuels and emissions-intensive materials in the energy and construction sectors. This is particularly the case for activities involving the enhancement of woodland productivity. Care may be needed to avoid woodland management or creation activities taking place on sensitive sites, or to ensure consistency of management with wider objectives (e.g. on sites of special scientific interest). Generally, some co-benefits should be supported and trade-offs avoided by ensuring that woodland management and creation is consistent with the existing UK Forestry Standard.

4.7.4.3 Magnitude

A quantitative assessment based principally on findings of Morison et al. (2012), Matthews (2020) and Matthews et al. (in prep) gives the following broad per-hectare estimates for the climate change mitigation potentials of woodland enrichment activities:

Estimates of climate change mitigation for woodland restoration, based on expert judgement, adjusted from potentials estimated for woodland creation and management activities, are in a range between -0.5 and -1.5 tC ha-1 yr-1 (-1.8 to -5.5 tCO2 ha-1 yr-1), depending on the local conditions and the period considered within a 100-year time horizon. If woodland "ecological reserves" are created, this mitigation comes entirely from woodland carbon sequestration; if woodlands are created for wood production, then a proportion of this potential is contributed by carbon sequestration in wood products and product substitution.

Woodland diversification activities can mitigate between -0.15 and -1.0 tC ha^{-1} yr^{-1} (-0.6 and -3.7 tCO_2 ha^{-1} yr^{-1}), depending on the activities being targeted to local conditions and the period considered within a 100-year time horizon.

The enhancement of woodland productivity can mitigate between -1.3 and -1.8 tC ha^{-1} yr^{-1} (-4.6 and -6.6 tCO_2 ha^{-1} yr^{-1}), with a proportion of this potential being contributed by carbon sequestration in wood products and product substitution.

The share of carbon sequestration benefits contributed above ground and below ground is variable and depends on the sites involved and the detailed woodland types and their management. It is likely to vary between a 50:50 share (above-ground:below-ground) for woodland restoration activities to an 80:20 share for activities aimed at enhancing woodland productivity. Soil types involved in woodland creation and management activities, and the management of those soils prior to implementation of the activities, will influence these shares.

4.7.4.4 Timescale

Based on the assessments in Morison et al. (2012), Matthews (2020) and Matthews et al. (in prep), it is estimated that:

- The carbon sequestration benefits of woodland restoration activities should start to be achieved within 5 years and can continue over several decades and possibly up to 100 years.
- The achievement of mitigation benefits through woodland diversification activities will be very variable over time, depending on the initial state of the woodlands involved and how the diversification is carried out. For example, there may be short-term losses of woodland carbon in some cases, such as when some trees are removed from woodland to encourage the regrowth of other tree species. In other cases, where tree species are introduced quickly in understocked woodlands, there are unlikely to be initial losses of carbon, but the main carbon sequestration benefits of woodland creation activities are achieved after 10 years. However, mitigation benefits can continue for up to 100 years or longer.
- Climate change mitigation benefits arising from activities involving the enhancement of woodland productivity are achieved after 10 years but the mitigation benefits can continue for up to 100 years or longer. Longer-term mitigation benefits come from increased supply of biomass for use in construction and for bioenergy, substituting for more energy-intensive materials.

4.7.4.5 Spatial Issues

The opportunities for implementing these actions in existing woodlands will be very specific to localities. For example, woodland restoration can only be undertaken in woodlands that are classified as 'degraded' in some way, whilst woodland diversification is appropriate where woodlands are not diverse already. Equally, some activities will only be appropriate in certain types of woodlands. For example, introducing faster-growing coniferous tree species is probably a suitable activity in some existing coniferous woodlands but it is highly unlikely to be suitable in existing broadleaf woodlands, for wider ecological reasons. These issues suggest a requirement for careful and sensitive opportunity mapping or site-by-site assessment before implementation.

4.7.4.6 Displacement

The activities take place exclusively in existing woodland areas so it is highly unlikely that agricultural production will be lost.

4.7.4.7 Maintenance and Longevity

Enhanced carbon stocks in woodlands achieved through these activities are potentially impermanent.
Support may need to be given to landowners and their agents undertaking woodland enrichment activities when their forest areas are affected by disturbance events outside of their control.
If circumstances require woodland enrichment activities to be modified or abandoned, then a mechanism is needed to ensure that compensatory actions are taken to conserve carbon sequestration and carbon stocks at target levels at the landscape/regional scale. These actions may involve the landowners involved or other agents.

When enhancing woodland productivity, climate change mitigation achieved through enhanced supply of woody biomass for construction and bioenergy, potentially substituting for more GHG-intensive materials and fuels, should be permanent.

4.7.4.8 Climate Adaptation or Mitigation

These activities are aimed at supporting a combination of climate change adaptation and mitigation goals, consistent with the concept of nature-based solutions.

Woodland restoration by definition should support woodland resilience to environmental change. Some consideration may need to be given to diversifying tree species as part of restoration, to ensure resilience to future climate change.

Diversifying woodlands should support the resilience of woodlands and spread risk from and the selection of tree species can and should take account of future climate change.
The selection of tree species when enhancing woodland productivity can and should take account of future climate change.

4.7.4.9 Climate Factors / Constraints

There are risks that future climate change mitigation benefits and co-benefits provided by woodland enrichment activities could be undermined if the climate becomes unsuitable for the tree species in the affected woodlands in the future. However, as already stressed above, these activities aim to address adaptation and resilience to climate change alongside providing these benefits.

4.7.4.10 Benefits and Trade-offs to Farmer/Land-manager

Assuming woodland enrichment activities are supported through financial incentives, farmers and land managers can potentially benefit from a more diversified source of income on their land and from resultant spread of business risk. However, the hosts of woodland enrichment activities would be required to maintain the woodlands in the improved state. Where the aim is to enhance productivity in woodlands, farmers and landowners may require support in developing markets and supply chain infrastructure to realise the benefits of increased availability of timber and biomass resources.

4.7.4.11 Uptake

Culturally, farmers can be uncomfortable with activities involving the maintenance and management of woodlands, which are not recognised as an agricultural activity and are not identified as a potentially viable source of income (Lawrence and Dandy, 2014).

A number of options are available to address this issue including direct communication, stakeholder participation in the development of the detailed methodologies and demonstration of good practice in a network of exemplar projects.

4.7.4.12 Other Notes

The participation by owners of small areas of land may need to be facilitated. One way of achieving this might be by providing a central scheme which can be joined by such owners and operated on a collective basis.

5 EVIDENCE GAPS

Evidence for the enhancement of carbon sequestration through management and habitat creation varies greatly in its scope and certainty across systems. Whilst evidence for increased carbon sequestration is strong in a number of places, we have identified evidence gaps that have the potential to limit the practical application of management actions for carbon sequestration.

5.6.1 Mapping carbon resources

Having a comprehensive baseline of national carbon stocks and their spatial distribution is key to reliable accounting and our ability to monitor changes in carbon stocks in response to management. Whilst current UK carbon accounting is performed to a high standard, evidence gaps in the distributions of some habitat types are known limitations. Going forward, nationally available LiDAR data and high resolution RGB imagery has the potential to supplement this information base, particularly when paired with field validation such as that provided by the Countryside Survey (Carey et al., 2008). For example, blue carbon has also been identified as a valuable carbon stock within the UK, however we lack accurate spatial data for coastal habitats at a national scale, on which to base coastal carbon accounting and as required to meet IPCC reporting standards (Parker et al., 2021). In the context of ELMs, confirming the added value of managements in terms of carbon sequestration is vital. Addressing this need in practice is likely to require a combination of on-site monitoring in and outside of ELMs sites, and use of ground-truthed, predictive models where measurements are not feasible.

The mapping of carbon resources, although facilitated and enhanced by remote sensing and modelling techniques, also requires a robust foundation in field monitoring data as discussed at the beginning of this report. However, robust monitoring of changes in carbon stocks is challenging for a number of reasons, including 1) the difficulty of detecting small changes in SOC against potentially large background stocks, 2) slow rates of change with the potential for significant lags between a change in management and an observable effect, 3) high spatial heterogeneity in resources at the scales of management implementation, 4) complex and varied baseline conditions, including management history, which can have strong legacy effects on observed changes in carbon stocks (Powlson & Neal, 2021). Some of these issues could possibly be addressed by the monitoring of suitable proxies for soil carbon change with more rapid response times to intervention. Powlson & Neal (2021) discuss the use of various proxies, including changes microbial biomass, but emphasise that these are unlikely to provide direct evidence for any absolute change in soil carbon stocks (Powlson & Neal, 2021).

5.6.2 Coastal and marine carbon stocks

In this evidence review we found highly unequal research effort across habitat types, with the majority focused on carbon stocks and fluxes in peat soils and woodland. On the one hand, these systems are significant players in national carbon sequestration potential, and their prioritisation is both logical and necessary. On the other hand, saltmarshes and sea grasses have also been identified as import drivers of carbon sequestration but comparatively little is known about their carbon dynamics, and even less is known about carbon dynamics in other systems, such as scrub habitats. Despite multiple case studies of salt marsh restoration leading to carbon stock accumulation (Burden et al., 2019; Parker et al., 2021), we sample sizes remain small and we do not understand the relative importance of deposition compared to in situ productivity, or rates of GHG emissions (particularly CH_4 and N_2O). Intertidal and subtidal sediments are also capable of storing significant carbon stocks. Although carbon dioxide is not being fixed from the atmosphere in situ in these systems, we have little understanding of the role of these systems as long term carbon stores, the sources of contributing sediment, and their vulnerability to disturbance. Whilst marine systems are outside the direct influence of land management actions considered in this report, coastal and marine systems are known to be sensitive to terrestrial inputs and disturbance from human activity.

5.6.3 Life cycle emissions

Most evidence for the impacts of management or habitat change on carbon stocks featuring in this review consider only the changes in semi-natural carbon stocks in situ. However, changes in management can also result in significant processing cost emissions or displacement (equation 1) which frequently go unaccounted for. This is particularly critical where expected rates of sequestration per hectare are low, but the area available for implementation makes those actions nationally significant, as a small increase in externalised emissions could transform assessments relatively easily. Anticipating emissions as a direct result of management actions is more feasible than accounting for the impact of displacement, due to the strong dependence of the latter on changing demand and international markets. However, the generalised effects of displacement are well known, and could have devastating consequences for carbon sequestration, particularly where this results in the conversion of carbon-rich semi-natural habitats to agricultural land, both nationally and internationally.

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