Woodlands for climate and nature:

A review of woodland planting and management approaches in the UK for climate change mitigation and biodiversity conservation

Report to the RSPB by Ellie Crane, February 2020

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Executive summary

General points

- There is frequently a negative relationship between the rate of carbon draw-down of a forest and the size and long-term permanence of its carbon stocks.
- Afforestation and forest management affect the climate through a variety of additional means including albedo, evapotranspiration, and aerodynamic surface roughness length.
- Changes to climate and nitrogen deposition rates will affect the future carbon balance of forests. Current models disagree on the magnitude and direction of this effect.

Forestry on mineral soils

- Management decisions involve trade-offs between ecosystem services including timber production, carbon sequestration and biodiversity. Any assessment of overall effects on biodiversity will similarly involve a subjective choice between species assemblages. Mapping approaches can facilitate spatial prioritisation of management actions.
- Summary of findings for carbon:
    - For fast carbon drawdown and high timber and biomass production, plant fast-growing conifers and harvest by clearfell soon after the trees reach their age of maximum growth. For large, long-term forest carbon stocks, plant slow-growing deciduous trees and manage under a low-intensity system such as continuous cover forestry or with a long rotation time.
    - Decide on thinning regime and harvest intensity (stem only versus whole-tree harvesting) on a case-by-case basis.
    - For resilience to climate change and its effects (including increased pests and diseases), increase the diversity of tree species and ages within stands.
- Summary of findings for biodiversity:
    - Protect old-growth semi-natural woodlands. Favour native broadleaved species for new plantations. Manage plantations (including by diversifying the species planted) so as to increase light levels and structural diversity and provide sufficient undisturbed and deadwood habitat.
    - There is a need for more studies that assess biodiversity more broadly at landscape scale and over the longer term.

Forestry on deep peat

Biodiversity

- Afforestation leads to significant loss of peatland biodiversity including on open peatlands adjacent to the forest plots themselves.
- Restored sites can make substantial progress towards ecological recovery in 10-20 years. Techniques are being developed to speed vegetation recovery.

Climate impacts of conifer plantations on deep peat

- It is generally accepted that new planting on deep peat has net negative impacts on ecosystem services including climate regulation. In the case of existing plantations, different studies have reached different conclusions on the net greenhouse balance of these afforested peatlands. However, the longer a plantation is left standing, the more carbon will be lost from the peat, and the harder it may be to eventually restore a functioning bog.
- Afforested peat soils experience ongoing loss of carbon. The rate of carbon draw-down by plantation trees can exceed the rate of carbon loss from the peat. However, the residence time of carbon in plantation trees and harvested wood products is much lower than in peat in an intact bog. Healthy peatlands store carbon and continue to absorb more from the atmosphere over periods of millennia. Therefore, any short-term gain in greenhouse gas balance from restocking may be at the expense of a much larger loss over the long term. It is important to note that forestry crops on UK peatlands are often of such poor quality that much of the wood goes for pulp, fuel and other short-lived uses.
- Drainage decreases methane emissions but may increase nitrous oxide emissions. Carbon is also lost from afforested peatlands in aquatic forms. The extent of the impact of plantations on the hydrology of adjacent peat has yet to be determined.

Climate impacts of restoring peatlands

- The evolution of greenhouse gas fluxes as restoration proceeds is complex and depends on, among other things, the conditions of the specific site prior to restoration and the particular restoration methods used. In general, restoration increases methane emissions and decreases carbon dioxide emissions. The net effect for climate mitigation partly depends on the timescale over which emissions are considered, as methane (although more potent per tonne than carbon dioxide) is a shorter-lived greenhouse gas. The climate benefits of restoration are greater when considered over a longer time period, as the importance of the methane emissions declines in relation to carbon dioxide emissions.
- Findings from recent research projects on Forsinard Flows are that restoration sites older than 15 years have a net climate benefit, taking into account carbon dioxide, methane and nitrous oxide. The results confirm the benefits of forest removal on deep peats where conifer yields have been low.
- Monitoring of restoration projects is providing data on the trajectory of vegetation recovery. Some studies have demonstrated that, where physical and chemical conditions are right, vegetation appears to be slowly returning to a more typical bog plant community. Remnant patches of original bog vegetation provide a source from which recolonization can take place. However, a variety of factors including elevated nutrient levels and dry patches can cause vegetation to succeed towards different plant communities.
- The emerging picture is that the ongoing sink function of bogs can be restored over a period of years, even before the bog vegetation makes a full recovery. New restoration methods are being developed and trialled, which could improve the timescales and

extent of recovery. All restoration management should take place in parallel with research and monitoring, within an adaptive management framework.

- Intact bogs are expected to be more resilient to the impacts of climate change than damaged bogs, but impacts of climate change on greenhouse balances are unknown.
- The fact that current understanding of the greenhouse gas fluxes of intact bog, plantations on deep peat, and restored peatlands is incomplete would seem to argue for a precautionary approach: protect the existing carbon store by protecting and restoring peatlands.

Forestry on shallow peat

- Shallow peat soils are likely to be vulnerable to carbon losses during the tree establishment phase and through erosion losses during ground preparation. A second period of vulnerability may be associated with forest harvesting both through physical disturbance and accelerated leaching losses of dissolved organic carbon.
- The current policy assumption is that over the long term, carbon losses from shallow peat are counterbalanced by gains from tree litter input, leading to no ongoing change in soil carbon stock. There is inconclusive evidence that this may be true over more than one forestry rotation. Further research is needed on the stability of this soil organic carbon.
- The biodiversity implications of afforestation on shallow peat are site-specific. Shallow peats often occur adjacent to or within a mosaic of deep peat and are hydrologically linked. Therefore, planting on shallow peat could also impact the hydrology, vegetation assemblages or priority species on nearby deep peat.
- It may not be appropriate to apply general models to specific sites based on the broad definition of "shallow peat".

Nature-based forestry

Minimum intervention

- In the UK, cessation of forest management has tended to lead to biodiversity loss. The extent to which biodiversity benefits from recommencing management depends on the specific types of management that are adopted.
- Policy to bring 'neglected' woods back into management is intended to increase carbon draw-down rates and provide a source of timber and fuelwood. However, resuming harvesting in unmanaged woodland would result in an initial decrease in carbon stocks, and even very old unmanaged forests can be large net annual sinks.
- 'Proforestation' - leaving forests intact and allowing them to grow to their ecological potential – has recently been put forward as a more effective climate change mitigation strategy than afforestation or reforestation.

Coppicing

- Coppicing (and other forms of traditional management) can provide the mix of living tree and deadwood habitats and semi-open habitats required by some species.

- Compared to high forests, coppiced woodlands are more resistant to water stress but have smaller carbon stocks.

Converting plantations to native woodland

- Partial or total conversion of large single-species conifer stands to native broadleaved trees can substantially increase native woodland species abundance and diversity.
- The process of conversion is likely to involve a short-term loss of carbon. In the longer term, conversion is likely to lead to a slower carbon draw-down rate but larger long-term carbon stocks, especially if harvest intensity is reduced.

Rewilding

- The biodiversity impacts of rewilding depend on the habitats present initially and once rewilding has commenced. Some claim that natural regeneration may be more beneficial for biodiversity than active afforestation.
- The carbon implications of rewilding similarly depend on the start and 'end' habitats. There is evidence that rewilding arable land can lead to significant carbon benefits, but the conversion of semi-natural grasslands to forest may not result in any significant overall gain in carbon stock.

Small-scale afforestation of arable land

- Evaluations of small woodlands in arable habitats have detected moderate benefits for taxa associated with open and young woodland habitats, but there is a lot of variation in woodland quality and characteristics of the surrounding landscapes.
- Woodland creation on arable soils will generally increase carbon stocks over the longer term, though carbon may initially be lost from the soil due to disturbance. The long-term carbon balance of the woodland created depends upon factors including species planted and how they are managed.

Conclusions

- There are few studies assessing the effect of afforestation on more than one ecosystem service at a time. This is a significant unmet need for evidence-based policy making.
- Managing forests for biodiversity may in general generate more synergies with maximising the long-term carbon store, and more trade-offs with maximising the rate of carbon draw-down.

Fate of harvested wood

- The fate of harvested wood is an important driver of the greenhouse gas balance of the overall forestry system. Harvesting reduces the equilibrium level of carbon in the forest but can provide long-term carbon storage opportunities outside the forest, as well as potentially reducing the use of fossil fuels and non-wood products.
- To gain an accurate picture of the carbon implications of harvesting forests, HWP and bioenergy life-cycle carbon analysis needs to be integrated with forest carbon balance analysis. Many previous studies have focused only on one or the other of these.

- Some authors have concluded that the no-harvest scenario is preferable to harvesting for bioenergy or HWP, i.e. that greater climate change mitigation can be achieved by leaving the forest standing and thus increasing its carbon stores. If harvest does take place, there is some consensus that long-lived HWP offer a more effective route to climate change mitigation than bioenergy. However, any significant shift from short-lived to long-lived HWP would present considerable challenges in terms of current production and consumption patterns. Reduction in or changes to UK wood production might result in increased imports from other countries, effectively offshoring the UK's footprint.
- Harvested Wood Products (HWP) represent a carbon store (although this is small compared to the carbon stock of the forest). Numerous studies have attempted to estimate residence time of carbon in different HWP, but there is significant uncertainty in these estimates. Both tree species choice and forest management affect the average lifespan of HWP. The carbon benefit of HWP can be enhanced by using more HWP in end uses with long service lives, increasing reuse and recycling of HWP, and using methane produced from decomposing HWP in landfills to generate energy.
- The use of HWP can reduce the use of carbon-intensive materials such as steel or concrete, leading to climate benefits from carbon displacement. However, estimates of the magnitude of the displacement effect vary widely and some authors have found that commonly cited figures are gross over-estimates. Furthermore, given that fossil fuels are a finite resource, any fossil fuels left unused by one sector may simply be taken up by other sectors in an effect known as 'carbon leakage'.
- Burning wood for energy releases carbon to the atmosphere. Unlike burning fossil fuels, this does not increase the total amount of atmospheric carbon in the long term. However, forest-based bioenergy cannot be considered carbon neutral because the payback time until the carbon is reabsorbed can be very long, particularly when living trees are felled for biomass. Harvest residues have a shorter payback time but increasing their use can have implications for the forest's continued ability to grow and absorb carbon. It is often argued that where carbon stocks are constant over a landscape scale (i.e. some forest stands are felled while others continue to grow) there is no carbon debt. However, this ignores the scenario where no harvesting is carried out, when the carbon equilibrium of the landscape would be higher.
- Replacing coal or gas with biomass for electricity generation is likely to significantly increase emissions per unit of electricity generated.
- By comparison, renewable technologies such as solar and wind power produce net carbon dioxide savings within months to a few years.

1 Introduction

1.1 Context to report

There is growing impetus to increase tree planting targets in response to climate change. As of 27 June 2019, the UK is committed to a target to reduce its greenhouse gas emissions by at least 100% (compared to 1990 levels) by 2050, also known as a net zero target (Priestly, 2019). The forestry sector is seen as having an essential role to play in meeting this target (CCC, 2019). Actions taken in the UK have far-reaching global consequences and there is an increasing need to ensure that woodland expansion is delivered in a way that delivers for biodiversity, the climate and other objectives. As stated by Burton *et al*. (2018): "*Research needs to make clear the effect of woodland expansion in different contexts, in order to provide robust, context-specific evidence. This is especially pertinent given the urgency of initiatives concerned with carbon sequestration and biodiversity protection, and the risk of rapid, poorly- informed actions leading to suboptimal or counterproductive outcomes*." The current report was commissioned by the RSPB to help inform its response to this need.

This report is based on a review of the peer-reviewed and grey literature, with additional input from specialists at the RSPB (see Appendix A for more detail).

1.2 Background: UK forestry

The area of woodland in the UK (as at 31 March 2018) is estimated to be 3.17 million hectares (Forestry Commission, 2018a). 83% of this area is managed for production (BEIS, 2016). UK woodland area has risen by around 250 thousand hectares since 1998, an increase of 9% (Forestry Commission, 2018a). Drivers to increase tree planting targets include the Clean Growth Strategy (HM Government, 2017) and recent recommendations on climate change adaptation through land use (CCC, 2018a) and the role of biomass (CCC, 2018b) from the Committee on Climate Change. The Committee has recently published a report presenting options for the UK to reach net-zero emissions by 2050 (CCC, 2019). One of the strategies presented involves afforestation of around 30,000 hectares per year to increase woodland cover from the current 13% of UK land area to 17% (actual afforestation was around 9,000 hectares per year between 2007 – 2017 (CCC, 2018b)). According to the Committee, this combined with an increase in active woodland management would increase the net carbon draw-down rate of forests in the UK to an estimated 22 MtCO2e per year by 2050 (CCC, 2019).

Approximately 1.6 million ha (51%) of the total UK woodland area is under conifer species (BEIS, 2016). Softwood (i.e. wood from conifers) makes up most of the wood harvested from UK forests. Scots pine is the only conifer species of economic significance native to the UK but Sitka spruce (native to North America) provides most of the timber for the wood processing industry (Forestry Commission, 2017). Sitka spruce accounts for around 51% of the conifer area in Great Britain, followed by Scots pine (17%) and Larches (10%) (Forestry Commission, 2018a).

Broadleaf forests include woodland managed primarily for conservation benefit, as well as commercial 'carbon offset' schemes, whereby woodland management organisations sell carbon credits to companies and individuals; and (generally) smaller-scale planting by individual landowners. The most commonly occurring broadleaved species in Great Britain are birch (accounting for 18% of broadleaf woodland), oak (16%) and ash (12%) (Forestry Commission, 2018a).

In theory, a minimum standard of sustainable practice is ensured through the UK Forestry Standard, which applies to all UK forests (Forestry Commission, 2017). In addition, the Woodland Carbon Code (introduced in 2011) is a voluntary standard for woodland creation projects that make claims about carbon sequestration. Further details are given at Appendix B. The UK Woodland Assurance Scheme is owned and managed by a broad partnership and is independent of government. It is based on the requirements of international forest certification schemes together with those of the UK Forestry Standard. The principal purpose of UKWAS is to act as an audit protocol for the independent certification schemes (Forestry Commission, 2017). Overall, 43% of the UK woodland area is certified (i.e. independently audited against the UK Woodland Assurance Standard) (Forestry Commission, 2018a).

1.3 Carbon fluxes of woodland

The most important greenhouse gas associated with woodlands is carbon dioxide (Matthews *et al*., 2014). Woodlands can produce emissions of nitrous oxide, which is a much more potent greenhouse gas than carbon dioxide, but is emitted in much smaller amounts. Woodlands on soils that are not saturated with water act as methane sinks (Sozanska-Stanton *et al.,* 2016). A small number of studies consider the effects of afforestation on greenhouse gases other than carbon dioxide, but the evidence is limited, site-specific, and hard to generalise (Burton *et al.,* 2018).

A simplified description of woodland carbon fluxes is as follows: trees and other vegetation absorb carbon dioxide from the atmosphere through photosynthesis. Simultaneously, carbon is released through respiration and from decaying wood and leaf litter. Plants accumulate carbon in their living tissues. This material may be harvested or die in situ. When plants or parts of plants die, the material joins the litter layer of the forest floor. This organic matter may decay and release its stored carbon back to the atmosphere, or it can be incorporated into the soil. Carbon can remain in the soil for varying amounts of time before being released as matter decays or becoming incorporated into stable mineral structures. The carbon stocks of woodlands are therefore in the living and dead biomass and in the soil.

In productive and fertile ecosystems, both plant production and litter decomposition are greater compared to unproductive ecosystems. The net result is that while productive ecosystems have a faster rate of carbon draw-down from the atmosphere (and may sometimes store more carbon above ground), they often also store much less carbon in the soil, and less carbon overall. Körner (2017) states that forest productivity is commonly negatively correlated with the carbon capital of a forest under a given set of environmental conditions. Furthermore, Díaz *et al.* (2009) state that there are fundamental physiological, evolutionary, and biogeochemical tradeoffs that prevent the simultaneous maximization of the rates of carbon flow, and the size and long-term permanence of carbon stocks. Climate change mitigation depends much more strongly on the amount and permanence of carbon in the biosphere than on how fast it is drawn down (Díaz *et al*., 2009), although some have argued that fast draw-down is of value as a way to 'buy time' while more long-term mitigation measures are put in place (IPCC, 2007). At any rate it is vital to make a distinction between the rate at which a forest draws down carbon from the atmosphere, and the size of the carbon stock in the forest. Terms like 'carbon sequestration' are sometimes used in a confusing way in the literature. In the current report, every effort will be made to use clear language such as 'the rate of carbon draw-down' and 'the size of the carbon stock' to avoid any confusion.

It is important to note that both afforestation and forest management affect the climate through a variety of means in addition to influencing the carbon cycle, including albedo, evapotranspiration, and aerodynamic surface roughness length. In some cases the net effect of these factors on the climate may outweigh the carbon effects (Burrascano *et al.,* 2016; Iordan *et al.,* 2018; Montenegro *et al.,* 2009; Naudts *et al.,* 2016). One study found that the net gain in forest area in Europe since 1750 created a carbon sink but decreased albedo, with the net effect of small summertime temperature increase (Naudts *et al*., 2016). The conversion of deciduous forests to coniferous forests across Europe has similarly resulted in changes in albedo, canopy roughness, and evapotranspiration from the land surface, which contributed to warming rather than mitigating it (Naudts *et al*., 2016). Another study found that afforestation does not cause net warming at the latitudes covered by the UK, although the authors note this was contrary to previous results (Montenegro *et al.,* 2009). A full discussion of this issue is beyond the scope of the current report.

1.4 Impacts of global changes

Climate change is expected to have a number of impacts on UK forests. At the basic level, while an increase in atmospheric carbon dioxide increases plant growth and thus carbon draw-down rates, an increase in temperature leads to a rise in the release of soil carbon (Goetz *et al.,* 2013). Predicted climate change impacts on the UK's forests include: soil moisture deficits limiting growth in some areas; higher productivity where soil water and nutrient availability allow; greater water table fluctuations over the year which will limit rooting depth and increase the risk of windthrow; higher incidence and severity of tree disease and pest outbreaks; and greater risk of fire (Ray *et al*., 2010, Forestry Commission research note).

In addition to climate change, changing levels of reactive nitrogen in the atmosphere have an impact on forests. Nitrogen deposition increased significantly from the late 19th to the early 21st century (Pretzsch *et al.,* 2018) but is expected to decrease over coming decades (Wamelink *et al.,* 2009). The stimulation of plant growth under elevated carbon dioxide is dependent on a sufficient supply of nitrogen. Under future climate conditions, nitrogen availability may have a major role in influencing carbon sequestration in forest ecosystems (Macdonald *et al.,* 2011). The impact of nitrogen deposition on carbon sequestration in forests is highly uncertain and may vary by two orders of magnitude (de Vries *et al.,* 2009).

Some models suggest that European forests could shift from net carbon sinks to net carbon sources in the 21st century (Herrero *et al.,* 2012; Goetz *et al.,* 2013). Other studies, however, find that under predicted climate change and nitrogen scenarios, carbon draw-down rates will increase across much of Europe's forest area (see for example Wamelink *et al.,* 2009). Valade *et al.* (2017) point out that uncertainty surrounding the future evolution of the forest sink under climate change and its interaction with management practices have mostly been ignored in modelling. As a result of this inadequate understanding, current models disagree on the magnitude and even the sign (sink or source) of the terrestrial carbon pool by 2100 (Valade *et al.,* 2017).

One study found that wood density of some tree species (Norway spruce, Scots pine, European beech and sessile oak) has decreased significantly since 1900 due to the changes in climate and nitrogen deposition (Pretzsch *et al.,* 2018). Lower wood density generally means a higher susceptibility to disturbance events such as high winds, which can reduce the carbon capacity of the forest. In terms of harvested wood, it means less calorific value per harvested volume (relevant for bioenergy) and inferior timber quality, which could result in more wood going to uses with shorter lifespans. This finding also means that less additional carbon has been sequestered due to increased growth rates than would otherwise be the case.

Some authors have gone so far as to say that storing carbon in forests may be an appropriate short- and medium-term mitigation policy, but to a lesser degree for certain forest ecosystems and locations in the long term (Goetz *et al.,* 2013).

2 Forestry on mineral soils

'Mineral soils' are defined as those with an organic layer of less than 5cm (Woodland Carbon Code, 2018). The current section of the report deals with forestry on mineral soils, although some points will apply equally to forests on peat soils. Issues that are specific to forestry on peat soils are discussed in a separate section.

Forest management affects carbon stocks in a variety of ways, discussed in the following sections. There are many interdependencies between the management choices discussed here, for example between the choice of tree species and rotation length (Jandl *et al.,* 2007).

2.1 Context: development of forest soils

Forest floors have a litter layer, beneath which is an organic layer and a deeper mineral layer (NB in the current report, 'mineral soil' is used to refer to the overall soil type of a site, while 'mineral layer' refers specifically to the soil horizon beneath the organic layer). Carbon is stored in and moves between each of these layers, in varying amounts depending on factors such as tree species, climate and soil type (Jandl *et al.,* 2007). Generally, the litter represents a small proportion of the forest's carbon stock, while the carbon stored in the soil layers often exceeds that in the trees (Matthews *et al*., 2014; Wamelink *et al.* 2009) (see also Table 1 in the current report). The *rate* of carbon accumulation following afforestation is, however, slower in the soil than in the above-ground biomass, because only a small proportion of plant-derived carbon becomes stabilised in the mineral layer (Jandl *et al.,* 2007). Nevertheless, modelling studies suggest that European forest soils are currently sequestering 26 Tg C yr^{-1}, which is 30–50% of the estimated carbon sink in the forest biomass (Grüneberg *et al.,* 2014).

Many forest carbon studies focus on the top 30cm of soil, even though significant amounts of carbon are known to be stored at greater depths (Ottoy *et al.,* 2017; Wiesmeier *et al.,* 2013), and trends in the topsoil carbon may not reflect trends in the mineral layer (Girona-García *et al*., 2018 and references therein; Callesen *et al.* 2015; Wiesmeier et al., 2013; Vesterdal *et al.,* 2013). Soil organic carbon is the major constituent of the soil carbon stock. Accurate assessment of soil organic carbon in forests is notoriously difficult due to very high spatial and temporal variability (Clarke *et al.*, 2015; Burton *et al*., 2018; Wiesmeier *et al.,* 2013; Hernández *et al.* 2017; Nave *et al.,* 2010).

When a site is to be afforested, site preparation is carried out to promote rapid establishment, early growth and good survival of seedlings (Jandl *et al*., 2007). This therefore promotes rapid growth of the above-ground carbon stock. Afforestation is generally accepted to increase above-ground carbon storage (Blanco, 2018; Jandl *et al.,* 2007). However, site preparation can lead to carbon loss due to disturbance of the soil, with the amount lost depending on the soil type and the specific site preparation techniques applied (Matthews *et al*., 2014; Burton *et al.,* 2018; Jandl *et al.,* 2007). Many site preparation techniques involve the exposure of the mineral soil by removal or mixing of the organic layer. This stimulates the decomposition of organic matter, leading to a net loss of soil carbon (Jandl *et al.,* 2007).

After the initial carbon loss following land use change, the soil begins to move towards a new carbon equilibrium. The magnitude and direction of the changes in soil carbon upon afforestation depends largely on the carbon stocks beforehand: for example, arable land tends to have low carbon stocks while pastures subject to less disturbance have large stocks (Matthews *et al*., 2014; Burton *et al*., 2018; Jandl *et al.,* 2007; Li *et al.,* 2012). A global meta-analysis found that when arable land is afforested, soil carbon stocks increase in the organic and mineral layers. However, when natural grasslands are afforested, soil carbon increases in the organic layer but decreases in the mineral layer, resulting in no overall change in total soil carbon stock (Li *et al.,* 2012; Burrascano *et al.,* 2016) (note this result does not include above-ground carbon).

Following afforestation, carbon accumulation initially takes place in the litter layer (Jandl *et al.,* 2007), but the evidence for effects on deeper soil layers is less clear. Studies in Denmark have found that litter carbon stocks were close to steady state 20–30 years after afforestation in deciduous forests on nutrient-rich, fine-textured soils, whereas litter carbon stocks continue to accumulate much longer on sandy soils and in coniferous tree species. On the other hand, mineral layer carbon stocks were found to decrease during the first two decades after afforestation and did not increase until three decades had passed since afforestation (Callesen *et al*., 2015 and references therein). Other studies have found that carbon gains in the upper mineral layer of plantation forests can be offset by losses of old carbon from deeper parts of the soil (Jandl *et al.,* 2007). Repeat surveys of soil carbon stocks in Danish forests in 1990 and 2008 found that soils subject to land- use change from cropland to forest during the study period accumulated carbon in the litter layers, but no change was detectable in the mineral layer. The carbon stock change in the overall soil profile was not statistically significant (Callesen *et al*., 2015). A study in Ireland found no significant difference in soil organic carbon concentration in the litter layer and top 30cm of soil between forested sites and their adjacent non-forest sites (Wellock *et al.,* 2011), but the authors postulate this may be due to an insufficient number of samples and high variability between sites.

In young plantations, litter inputs tend to be smaller than soil organic matter decomposition (Blanco, 2018 and references therein; Jandl et al., 2007). The age at which a plantation transitions from a net carbon source to sink is variable even among the same tree species and depends on climate, soil characteristics, site management and disturbance history (Chan *et al.,* 2018). It may take decades until net soil carbon gain is achieved, even on former arable soils (Blanco, 2018 and references therein; Jandl et al., 2007). One review of field studies concluded that, following the initial loss upon afforestation, soil carbon tends to recover or increase by the second rotation (Burton *et al*., 2018). Where the initial soil carbon stock is very high (as is the case for peatlands), the loss upon afforestation may be so high that there is never a complete recovery of soil carbon (Matthews *et al*., 2014). Forest soil surveys in Denmark revealed that soils with moderate initial carbon contents gained carbon between 1990 and 2008 whereas very carbon-rich and organic soils lost carbon (Callesen et al., 2015) (NB the forests surveyed were of varying ages).

The general consensus seems to be that, except on peat soils, afforestation is likely to be beneficial for soil carbon stocks on the whole (Burton *et al*., 2018; Matthews *et al*., 2014; Jandl *et al.,* 2007).

It is important to make a distinction between carbon accumulation in soil and carbon stabilisation (Blanco, 2018; Jandl *et al.,* 2007). Carbon accumulation simply means that inputs are greater than outputs – this could arise, for example, because of a large input of material from brash left after harvest, or because of reduced soil respiration due to climatic conditions. Forest floors accumulate carbon quickly, but most of it is in a labile form and residence time is short (Jandl *et al.,* 2007). Carbon stabilisation involves carbon forming complexes with other elements of the soil, effectively locking it in even if site conditions change, for example if the forest is clearfelled (Blanco, 2018; Jandl *et al.,* 2007). Carbon accumulation can contribute to climate mitigation in the short term, but carbon stabilisation is essential to create a long-term carbon store. The rate of carbon stabilisation is largely determined by soil properties, for example clay content and whether the soil is derived from calcareous or non-calcareous parent material (Jandl *et al.,* 2007; Grüneberg *et al.,* 2014).

Table 1: size of carbon stocks in different forest compartments

Summary of data extracted from literature review

<table>
<tr><th rowspan="2">Reference</th><th rowspan="2">Case study details</th><th colspan="3">Size of C store (in tonnes C per hectare)</th></tr>
<tr><th>Above-ground</th><th>Litter</th><th>Below-ground</th></tr>
<tr><td>Jonard et al. (2017)</td><td>Sampling of forest sites across France. To 1m depth.</td><td></td><td>25.4 (pine)
16.1 (N. spruce)
10.7 (Douglas fir)
5.7 (oak, beech)</td><td></td></tr>
<tr><td>Nijnik *et al.* (2013)</td><td>Beech plantations. Above ground only (averaged over length of rotation).</td><td>1.9 – 4.8 (unthinned)
1.5 – 3.9 (thinned)</td><td></td><td></td></tr>
<tr><td>Ottoy *et al.* (2017)</td><td>SOC of mixed forest in Belgium. To 1m depth.</td><td></td><td colspan="2">175.8</td></tr>
<tr><td>Lettens *et al.* (2005)</td><td>SOC of forests in Belgium</td><td></td><td>10 (broadleaf)
20 (mixed)
35 (coniferous)</td><td></td></tr>
<tr><td>Lettens *et al.* (2005)</td><td>SOC of forests in Belgium. To 1m depth.</td><td></td><td></td><td>148 (broadleaf)
155 (coniferous)</td></tr>
<tr><td>Grüneberg *et al.* (2014)</td><td>National soil inventory for German forests. Organic layer only.</td><td></td><td colspan="2">18.8 (overall average)
14–26 (coniferous)
10–20 (mixed)
5–11 (deciduous)</td></tr>
<tr><td>Grüneberg et al. (2014)</td><td>National soil inventory for German forests. First 30cm of mineral layer.</td><td></td><td></td><td>61.8</td></tr>
<tr><td>Prada *et al.* (2016)</td><td>Sweet chestnut coppice stands, northern Spain. Aboveground plus organic layer and first 1m of mineral soil</td><td>119.75</td><td colspan="2">179</td></tr>
<tr><td>Lee *et al.* (2018)</td><td>Oak coppice in Turkey</td><td colspan="3">116.0 - 140.3 (coppiced)
128.1–236.2 (abandoned)</td></tr>
<tr><td>Brainard *et al.* (2009)</td><td>55 year old oak/ beech forests in Great Britain. Live vegetation only.</td><td>59.7 - 85.0</td><td></td><td></td></tr>
<tr><td>Nijnik et al. (2013)</td><td>Sitka spruce stands. Above ground only (averaged over length of rotation).</td><td>2.2 – 6.5 (unthinned)
1.9 – 5.3 (thinned)</td><td></td><td></td></tr>
<tr><td>Gielen *et al.* (2013)</td><td>80-year-old Scots pine forest in Belgium. Above-ground and to 30cm depth.</td><td>94</td><td>30</td><td>84</td></tr>
<tr><td>Brainard *et al.* (2009)</td><td>25- 40 year old stands of Sitka spruce in Great Britain. Long term equilibrium storage in trees and litter.</td><td colspan="2">73.2 – 82.0</td><td></td></tr>
</table>

Reference	Case study details	Size of C store (in tonnes C per hectare)		
		Above-ground	**Litter**	**Below-ground**
Wiesmeier *et al.* (2013)	Analysis of soil sampling from forests across Bavaria (predominately mineral soils but some organic soils). To at least 100 cm depth.			94 – 99
Wiesmeier *et al*. (2013)	Analysis of soil sampling from forests across Bavaria (predominately mineral soils but some organic soils). Organic layer (O) plus the first horizon of the mineral layer (M).			Broadleaf: 6 (O) + 41 (M) = 47 Mixed: 18 (O) + 26 (M) = 44 Conifer: 32 (O) + 14 (M) = 46
Wellock *et al.* (2011)	Irish forest soils. To 30cm depth.		1.7 (broadleaf) 3.2 (mixed) 9.3 (conifer)	

2.2 Tree species choice in commercial forestry

Many different tree species are grown commercially in the UK. The species present in a forest, and in particular whether they are coniferous or broadleaved, has numerous effects on carbon fluxes and biodiversity.

2.2.1 Soil carbon pools

The organic layer and upper horizon of the mineral layer are strongly influenced by tree species via litterfall, whereas any impacts on the deeper mineral layers are less direct and consistent (Girona-García *et al*., 2018 and references therein; Wiesmeier *et al.* 2013; Jandl *et al.,* 2007; Vesterdal *et al.,* 2013). This finding may be partly because of lack of applicable research: some authors have cautioned that changes to the mineral layer carbon stock will take longer to appear following a change in tree species, and that few long-term studies exist on this topic (Vesterdal *et al*., 2013).

The authors of one review found that pine forests have low soil carbon pools, whereas beech forests have the highest soil and total carbon pools (Jandl *et al.,* 2007). They point out, however, that mean values for different species also represent site conditions where the species are dominant. For instance, Scots pine forests often grow on shallow and dry soils, which have low carbon stocks, whereas beech is found on more fertile soils (Jandl *et al.,* 2007).

Some authors have concluded that the evidence points to tree species influencing the distribution of carbon in the soil profile, rather than the total carbon stock (Vesterdal *et al*., 2013). Coniferous forests tend to have thicker litter and organic layers than deciduous forests, because the litter produced by coniferous trees is slower to decompose (Girona-García *et al.,* 2018; Wiesmeier *et al.,* 2013). This leads to larger carbon stores on the forest floor and surface soil, which may partly explain observations that coniferous stands have the capacity to accumulate soil organic carbon at a higher rate than broadleaved forests (Jonard *et al*., 2017 and references therein). Studies in Spain have explored the effect of converting native beech forests to Scots pine plantations (Girona-García *et al*., 2018 and references therein). One study found that, 51–128 years after conversion, there was a 75% increase of soil organic matter in the organic layers of pinewoods compared to the original beech forests. A study of 21 afforested sites in Ireland found that coniferous forests stored significantly more carbon in the litter layer than mixed or deciduous forests (Wellock *et al.,* 2011) (Table 1). One review of carbon stocks in the litter layer of temperate and boreal forests ranked broadleaf species as follows (from high to low): beech, oak, followed by ash, maples, sycamores and lime (Vesterdal *et al*., 2013). Earthworm abundance is proposed as one likely explanation for the different distribution of soil carbon stocks under different broadleaf species. Conifers (spruce, hemlock and larch) had litter layer carbon stocks at least as high as beech (Vesterdal *et al.,* 2013).

A global meta-analysis of temperate forests found that coniferous and mixed forests lose less soil carbon from the litter layer when harvesting takes place than do deciduous forests, possibly due to the greater recalcitrance of coniferous residues (Nave *et al.,* 2010). When considering only the mineral layer, soil type was found to be the most important predictor of harvest impacts on soil carbon. Tree species did have some influence, with coniferous and mixed forests showing no significant change in mineral layer carbon storage following harvest while hardwoods lost about 9% of their mineral layer carbon. Carbon losses from the mineral layer were detected 0.5 years post-harvest, but recovered after 6 – 20 years (Nave *et al*., 2010).

Conifers are shallow-rooting and, although as discussed above they tend to accumulate more carbon in the shallow soil layers, they accumulate less in the mineral soil layers compared to deciduous trees (Jandl *et al.,* 2007). Broadleaved trees have a greater capacity to transfer carbon to deeper soil layers and stabilise it in the mineral layer (Wiesmeier *et al*., 2013). Broadleaved forests often have higher root biomass than conifers, and it has been shown that root litter may contribute as much carbon to the soil stocks as foliar litterfall (Vesterdal et al., 2013). Root-derived carbon may also be more stable than foliar litter-derived carbon in certain ecosystems (Vesterdal *et al.,* 2013). In a Spanish study (Girona-García *et al*., 2018 and references therein), conversion from beech to pine led to a 50% decrease in soil organic carbon in the soil mineral layers down to 60 cm depth. Another study of beech and pine forests in Spain down to 100 cm depth found that soil organic matter incorporated into the mineral layer under the beech forest was more stable and well preserved compared to the pine forests (Girona-García *et al*., 2018 and references therein). Repeat soil surveys in German forests 16 years apart found a decrease in carbon stocks of the organic layer under deciduous trees, but an increase in the mineral layer carbon pool (Grüneberg *et al.,* 2014). A global meta-analysis of afforestation found that carbon stocks in the organic layer were increased for pine, softwoods and hardwoods, with the stock in the pine plantation significantly higher than in the others. For the mineral layer, on the other hand, soil carbon stocks decreased for pine, and increased for hardwoods. Total soil carbon increased for hardwoods but did not change for softwoods including pine (Li *et al*., 2012). In a meta-analysis of soil sampling from forests across Bavaria, Germany (Wiesmeier et al., 2013), no significant differences were found in total soil organic carbon between deciduous, mixed and coniferous forests. However, when the different soil layers were considered separately, broadleaf forest soils had low carbon stocks in their organic layers and high carbon stocks in the uppermost horizon of their mineral layers, while conifers had the opposite trend (and mixed forests had intermediate values) (Table 1). That is, in coniferous forests a higher proportion of the soil carbon stock was in labile forms near the surface. There were no significant differences in carbon stocks between forest types in the deeper mineral horizons. The authors of this study acknowledge that many earlier studies had found that soil organic carbon is higher under coniferous forests than deciduous forests, but point out that most of these studies did not account for carbon in deeper soil layers (Wiesmeier *et al*., 2013).

Carbon stores in the litter and shallow soil layers are vulnerable to disturbance (Wiesmeier *et al.,* 2013). If the objective of afforestation is to create long-term carbon stocks, it is important to select tree species that favour the stabilisation (not just the accumulation) of soil carbon (Blanco, 2018; Wiesmeier *et al.,* 2013; Jandl *et al.,* 2007). Wiesmeier *et al.* (2013) go so far as to say that, under conditions of climate change, the maintenance of large coniferous forest areas could lead to a notable decrease of soil organic carbon and turn forest soils into carbon sources instead of sinks. It has been suggested that planting mixed stands may be a way to combine the advantages of conifers (high rates of carbon accumulation in litter and organic layer) and deciduous trees (improved stabilisation of carbon in long-term stocks in the mineral layers) (Jonard *et al*., 2017 and references therein). However, the question as to whether increasing tree species diversity per se will affect soil carbon stocks has yet to be answered (Vesterdal *et al.,* 2013). One study modelled the introduction of deciduous tree species to Norway spruce plantations in Austria (Seidl *et al.,* 2008). The results showed only a moderate response of overall forest carbon stocks over 100 years, with considerable uncertainty attached to the figures. However, a significant increase in belowground carbon stocks was predicted.

2.2.2 Above-ground carbon pools

In general, conifers produce stem volume more quickly than broadleaf trees. However, broadleaf tree species tend to have more branchwood than conifers and their wood tends to be higher density. The total aboveground accumulation of carbon for conifer and broadleaf species is therefore often very similar (Jandl *et al.,* 2007).

In a UK context, if the aim is to create a large and long-lasting carbon stock, then maintaining old-growth stands of slow-growing broadleaf tree species will be most effective. If on the other hand the aim is to manage forests for high levels of timber and biomass production (to achieve carbon substitution in the energy and construction sectors), then rapidly growing pioneer conifer and broadleaf species will generally be used (Matthews *et al*., 2014; Burton *et al.,* 2018). The latter approach is expected to be less effective at increasing carbon storage over the long term, since carbon stocks in harvested, fast-growing forests have short residence times (Burton *et al.,* 2018; Körner, 2017), and harvested wood products only contribute to reducing atmospheric carbon if the total size of the carbon pool stored in them increases over time (Burton *et al.,* 2018). In Northern Spain, there has been a widespread replacement of native deciduous forests with exotic pine and eucalyptus plantations, managed by rotational clearfelling to supply the timber industry. This has led to loss of biodiversity and declines in soil and water quality, but moves to restore native, slow-growing trees are often countered with the argument that fast-growing plantations are a more effective climate change mitigation measure (Rodríguez-Loinaz *et al.,* 2013). A study was carried out to test this assumption by modelling changes in the carbon stock in living biomass over 150 years under different scenarios. It was found that replacing pine and eucalyptus plantation in areas of high erosion risk with native broadleaves would lead to smaller carbon stocks in living biomass over the first 50 years, but larger stocks thereafter. The volume of commercial timber production would however be reduced (although the wood that was produced would be of higher value) (Rodríguez-Loinaz *et al.,* 2013).

It is important to consider the need to meet demands for wood products. The UK is currently a net importer of timber, and any decrease in domestic production could simply result in increased imports, shifting the impacts on forest carbon stocks to other countries (Burton *et al.,* 2018; Brainard *et al.,* 2009). There is therefore a role for efficient management of faster growing tree species, alongside protection and maintenance of old-growth stands, within climate change mitigation strategies (Burton *et al.,* 2018).

Even-aged coniferous forests at sites which would naturally support deciduous forest are prone to an array of diseases and pest organisms and may be particularly vulnerable to warmer and drier climatic conditions (Seidl *et al.,* 2008). Species diversification at stand and landscape scale is recommended as a measure to increase forestry productivity and resilience in the face of climate change (Ray *et al.,* 2010; Matthews *et al*., 2014; Wiesmeier *et al.,* 2013; CCC, 2018b; Barsoum *et al*., 2016).

2.2.3 Biodiversity

A review of the evidence base for the effects of woodland expansion on biodiversity (Burton *et al.,* 2018) found evidence for a number of impacts on biodiversity of afforestation with non-native conifers. A woodland flora has been observed to develop in some stands over time, but this is dependent upon a variety of factors including stocking density, previous land use and proximity to existing woodland. For invertebrates and birds, open ground species decrease at the expense of generalist and forest specialists. Diversity in forest structure and species is essential for maintaining ground beetle (Carabidae) diversity. There is strong evidence for edge effects of conifer plantations negatively affecting a number of open ground specialist bird species, particularly in upland contexts. Pre-afforestation land use is shown to have an influence on both bird and spider communities after coniferous afforestation. Improved grasslands are most likely to benefit from afforestation in biodiversity terms, whereas wet grasslands and peatlands are more sensitive (Burton et al., 2018).

Sitka spruce trees are particularly efficient at light interception, with the result that the understorey is almost entirely eliminated as the stand matures (Saraev *et al.,* 2017, Forest Research report). However, if managed with continuous cover systems allowing trees to mature beyond commercial clearfelling age, they can achieve more diverse stand structure with understorey and greater diversity of birds (Calladine *et al*. 2015, Currie & Bamford 1982). A review of UK forestry practices found that the largest differences in ground flora were between tree species that had dense canopies and species with more open canopies. This difference was much greater than differences between conifer and broadleaf plantations (Bellamy & Charman, 2012).

Burton *et al*. (2018) found that there is a lack of controlled, field-based evidence for the effect of native woodland expansion on biodiversity. Studies suggests that earthworm communities are larger under re-established native woodland than surrounding moorland, whereas studies of moths in recently-afforested sites found lower species abundance and richness compared to mature woodland. GIS-based work suggests that targeting new native woodland adjacent to ancient woodland patches increases core habitat area and functional network size, enabling faster colonisation of woodland species. The authors note that most studies focus on birds, invertebrates and ground flora, highlighting evidence gaps for the effects of new woodland on other taxa including mammals (Burton *et al*., 2018).

A study in Czechia compared biodiversity in non-native Norway spruce plantations with areas dominated by the native sessile oak (Horák et al., 2019). The number of species found was significantly higher in oak stands than in spruce stands, but there was no significant difference in the incidences of red-listed species between oak and spruce stands. Beetles and birds were negatively affected by increasing proportions of spruce in the tree composition, but unexpectedly three groups (mycorrhizal fungi, tree seedlings and bees and wasps) were positively influenced by increasing spruce dominance; and in fact the total number of species that responded positively to spruce was higher than those that were adversely affected. Many other species were not affected either way by the dominance of spruce. The authors concluded that, while the retention of native tree patches is critical, mixed species plantations (even if the species are non-native) can also benefit biodiversity. The UK Forestry Standard advocates a diversification in forest composition so that no more than 75% of a forest management unit is allocated to a single species and at least 5% comprises native broadleaved trees and shrubs (Forestry Commission, 2017). There are however concerns that this requirement is not sufficiently progressive to meet UK biodiversity obligations, and about its implementation in the absence of the auditing and monitoring processes which apply to the UK Woodland Assurance Standard (Neil Douglas, RSPB, personal communication July 2019).

The understorey plant communities of mature Sitka spruce, Norway spruce, Japanese larch and ash plantations were compared with those of semi-natural oak and ash woodlands in a survey of 75 sites across the island of Ireland (Coote *et al.,* 2012). The authors conclude that plantations of broadleaves and conifers have the potential to support high numbers of woodland species, and plant communities similar to semi-natural woodland. They recommend that to improve the net effects on biodiversity when afforesting areas adjacent to semi-natural woodland or in areas with historic woodland cover, broadleaved species, preferably native broadleaves suited to the soil type, should be chosen. Conifer plantations already present should be considered for conversion to native tree species or a native/non-native mix. For those forests planted in historically or currently unwooded areas, the number of woodland species supported can still be enhanced through maintaining adequate below-canopy light levels by planting of broadleaves or open canopied conifers, alone or in a mixture (Coote *et al*., 2012).

A study in Sweden compared the bird species composition and diversity of planted production oak stands with oak- dominated protected forest areas (Felton *et al.,* 2016a). The results indicated that mature production stands possessed a bird community partially overlapping in species composition (including red-listed species), and comparable in species richness to that found within the natural stands. There were, however, significant differences (for example the rarity or absence of species that nest in tree-hollows or rely on sun-exposed structures in production forests). The authors conclude that while production oak stands cannot replace the habitat provided in old-growth oak-dominated forests, they can supplement it, especially if sympathetic management practices are adopted. Similar to the Irish study (Coote *et al*., 2012), they recommend that broadleaf species should be used for new plantations in this region due to their potential to provide similar habitats and resources to the natural forest cover.

An analysis of Swedish forest and environmental policy concluded that converting native conifer stands to native broadleaved species would bring multiple benefits for commercial forestry (Felton *et al*., 2016). As well as biodiversity benefits related to the provision of deadwood and diversified forest structures, it would be a potential means of reducing the risk of wind damage and pest and pathogen outbreaks.

A review of UK forestry practices (Bellamy & Charman, 2012) concluded that the choice of tree species can have an effect on biodiversity but that the effect will depend on the taxa of interest. For example, for birds there was no difference in species richness or abundance between plantations of different tree species, although there were distinct communities for different tree species. This review also found that mixtures of tree species did not add any value to biodiversity other than that associated with the individual tree species.

The above brief review leads to the following conclusions:

- Plantations managed primarily for timber production have the potential to support significant levels of forest biodiversity, if managed appropriately (e.g. retention of dead wood, mix of tree species rather than monoculture).
- However, some specialist fauna species require old-growth forest and/or native tree and shrub species, so retention of semi-natural woodland is essential: plantations can supplement but not replace the habitat provided by old-growth forests.
- Choosing native broadleaved species for new plantations will benefit biodiversity (and may have additional benefits e.g. resilience to pests), particularly where there are existing patches of semi-natural woodland close by or there was historic woodland cover.
- To increase the biodiversity of closed-canopy conifer plantations (such as Sitka spruce), increase structural and species biodiversity by introducing broadleaved species or open-canopied conifers.

2.2.4 Novel species

It has been proposed that a range of novel exotic tree species should be grown in the UK in order to maintain forest productivity under climate change, create a more diverse and resilient forest, and substitute for native species threatened by introduced pests and pathogens (Ray *et al*., 2010; CCC, 2018b; Ennos *et al.,* 2019; Silvifuture, *online*). Based on a review of experiences in the UK to date, Ennos *et al*. (2019) concluded that where production is the main objective, there are strong arguments for undertaking a programme of rigorous testing and domestication of a small number of the most promising novel exotics. The species chosen must have attributes that will allow the development of more naturalistic silvicultural systems and a move away from current clear-fell regimes (as well as good timber and growth). The authors caution that testing must be undertaken within a comprehensive risk assessment framework (Ennos *et al*., 2019).

Others have concluded that planting non-native tree species is likely to lead to negative sustainability impacts (CCC, 2018b). The risk of a species becoming invasive is location-dependent. Poplar and eucalyptus, for example, have both shown invasion (CCC, 2018b). Where conservation of biodiversity is an objective, Ennos *et al.* (2019) find no support for introduction of any non-native species. Use of non-natives is likely to lead to an increase rather than a decrease in pest and disease problems, and to hinder rather than support the retention of threatened native tree species and their associated biodiversity.

In Sweden, most production forests are currently native conifers. However, the introduction of non-native species has been suggested as a climate change adaptation measure. A study evaluating Sweden's environmental strategies concluded that the introduction of non-native species could reduce the benefits of conservation efforts (such as dead wood and green tree retention in harvested stands) because the introduced species do not provide the resources needed by native fauna. Conversely, where introduced species have a less dense canopy than native species, they can support a relatively rich understorey plant community. Introducing new tree species may also contribute to the structural diversity of forests if they are harvested at different age to the native trees (Felton *et al*., 2016).

2.3 Rotation length in clearfelled plantations

Historically, rotation lengths in commercial forestry have been selected to maximise site productivity (Sing *et al*., 2018) and/or profitability. However, society increasingly expects forestry to deliver a variety of benefits, and attempts are being made to incorporate other factors into calculations of optimum rotation lengths (Saraev *et al.,* 2017, 2017a, 2019). In the UK, forest rotations last between 40 and 60 years for conifers, whereas for broadleaves they are typically 80 to 100 years (Whittaker *et al.,* 2011). Because of the increase in Sitka spruce growth rates that have been achieved through tree breeding and silviculture, trees are able to reach a merchantable size at younger ages and therefore rotation lengths are generally declining. Currently, a typical rotation length for commercial Sitka spruce in Great Britain is 35–45 years (Moore, 2011).

2.3.1 Soil carbon pools

Rotation length affects the amount and type of litter reaching the forest floor. The biomass allocation of the tree species present (stem versus non-stem) and how this changes with tree age interacts with rotation length to determine impacts on litter fall (Kaipainen *et al.,* 2004).

According to modelling of a rotational Sitka spruce plantation, generated using the CARBINE forest carbon accounting model, clearfelling is observed to result in a short-term reduction in soil carbon stocks, followed by a lower rate of carbon sequestration in soil compared to the case involving no clearfelling (Matthews *et al*., 2014). Repeated rotations may result in decreasing productivity and soil quality over longer periods (references within Seidl *et al.,* 2008).

Also according to the CARBINE-based model, carbon stocks in litter can be greatly increased by tree harvesting when material from felled trees is left on site, but the litter decays quickly, making only a short-term contribution to overall carbon stocks (Matthews *et al*., 2014). In one modelling study of Scots pine in Finland, under longer rotations the large tree biomass produced a lot of litter, but there were less harvest residues because of the decreased harvest frequency and because the larger trees left less residues per harvested volume (Kaipainen *et al.,* 2004). The net result was that soil carbon was relatively insensitive to rotation length.

Some modelling studies suggest that extending the rotation length may not increase the total carbon balance of the forest if the existing harvest age of the trees is beyond the stage of maximum annual growth. In older stands, productivity and therefore litter production decline, so the rate of soil carbon accumulation may level off (Jandl *et al.*, 2007). Also, over-mature even-aged stands (i.e. those that have been left standing beyond their age of maximum annual growth) are slower to close canopy gaps created by natural mortality or thinning, so the impact on soil carbon of these events is of longer duration (Jandl *et al*., 2007). Kaipainen *et al.* (2004) looked at Sitka spruce in the UK and Norway spruce in Germany. Results showed that the average carbon stock of soil increased with increasing rotation length up to a rotation of 80 or 90 years but not further, but the total carbon stock of the forest continued to increase with further increase in rotation length, following the trend in the carbon stock of trees (Kaipainen *et al.,* 2004).

The formation of stocks of stabilised soil carbon requires time, and therefore minimising soil disturbance (including from harvest) is important for long-term carbon storage (Jandl *et al.,* 2007).

2.3.2 Above-ground carbon pools

In forest stands subject to rotational clearfelling, carbon stocks in trees accumulate from time of planting up to the end of the rotation, when clearfelling effectively reduces carbon stocks in living trees to zero. The carbon stocks then accumulate again following replanting (as presented by Matthews *et al*., 2014 in a modelling exercise). Allocation of carbon to dead wood pools increases with stand age (Nunery et al., 2010; Luyssaert et al., 2008).

Rotation lengths where the time of harvest is close to the age of maximum tree growth will maximize aboveground biomass production (Jandl *et al.,* 2007). However, extending the rotation beyond this results in higher long-term average forest carbon stocks, both due to the relationship between tree size and duration of growth during the full vigour phase (Sing *et al.,* 2017) and because a longer rotation decreases the annually harvested area, thus increasing the average carbon stock of the trees (Kaipainen *et al.,* 2004). In a study modelling conifer forests in Europe (Kaipainen *et al*., 2004), the carbon stock of the biomass in all forests increased when the rotation length was increased. A 20-year increase in current rotation lengths increased the average carbon stock of biomass in pine forests by 6–13% and in spruce forests by 14–67% (Kaipainen *et al.,* 2004). Sitka spruce plantations in the UK were one of the forests considered. Increasing the rotation length from 40 years to 60 years increased the carbon stock in the trees by an estimated 24.6 tonnes per hectare. Added to an increase in soil carbon stock of 11.7 tonnes per hectare this led to an overall increase of forest carbon stock of 36.3 tonnes per hectare (Kaipainen *et al.,* 2004). The authors noted that in the Sitka spruce forests the current rotation length was so short that the production of the forests remained high even after the rotation length was increased (Kaipainen *et al.,* 2004). Modelling of northern hardwood-conifer forests in the United States (Nunery *et al*., 2010) showed that management practices favouring lower harvesting frequencies and higher structural retention sequester more carbon above ground than intensive forest management even when the carbon stored in harvested wood products is taken into account (substitution effects were not included in this study).

It is therefore generally accepted that longer rotations result in a larger above-ground stock of carbon. There are also economic arguments for extending rotations. The sale of carbon 'credits' on the voluntary carbon market is established in the UK. A recent Forestry Commission research note (Haw, 2017) concludes that even at low carbon prices, the extra carbon revenue generated from increasing commercial rotation lengths by five years outweighs the reduction in timber value from delayed harvesting. According to further economic modelling carried out for Forest Research (Saraev *et al.,* 2018a), an increase in timber prices would shorten the economically optimum rotation length for Sitka spruce plantations on sites with low windfall risk. However, an increase in the value assigned to carbon sequestered in the forest would extend the rotation length. Using input figures based on published values, the classic economic model (which only considers timber value) produces an optimum rotation of 42.7 years for Sitka spruce. The authors of the new model show that this increases to 50.5 years if the value of carbon stored in the forest is taken into account. However, at sites where wind damage is a significant risk, optimum rotation is only 40.8 years (Saraev *et al.,* 2018a). Given that actual rotations for Sitka spruce can now be as short as 35 years (Moore, 2011), it appears that the carbon stocks of forests are currently undervalued by the forestry industry. One explanation may be perceived or actual increased risks of natural disturbance associated with higher carbon stocks (see for example Matthews *et al*., 2014). The reasons put forward by these authors are that single-aged stands with large carbon stocks represent more of a fuel source for fire, big trees are more prone to storm damage than small trees, and older trees may be more susceptible to attack by certain diseases. Other factors may include timber prices and desire to realise assets within owner lifetime.

2.3.3 Biodiversity

Plantations have been shown to harbour both early-successional and late-successional species of conservation value. This suggests rotation times are an important management consideration in conserving such species (Calviño-Cancela *et al.,* 2012).

Younger woodland stages harbour bird species typical of open habitats, while species assemblages in older stages are characterised by woodland generalist species (Burton *et al.,* 2018). Some studies indicate that availability of young woodland is particularly important for several bird species (Burton *et al.,* 2018). For birds and mammals, a review of evidence carried by Forest Research suggests that for the first few years after a stand begins to grow, there is an increase in species richness and relative abundance (Saraev *et al.,* 2017). Once the canopy closes, habitat structure becomes unsuitable for open-country species, whilst initially not providing adequately for woodland specialists (though still providing breeding habitat for some generalist woodland birds) (Bellamy *et al.,* 2004; Whytock *et al*., 2017). When the lower branches of the trees meet and interlace after canopy closure (referred to as the 'thicket stage' and occurring at 10 – 20 years in conifer forests), species richness and abundance both fall, reaching a minimum at around 20 years (Saraev *et al.,* 2017). They then increase again, although some of the increase is only obtained at ages beyond typical rotation lengths (Saraev *et al.,* 2017). It may take 50-100 years for woodland to provide the requirements of more specialist woodland bird species (Bellamy *et al.,* 2004; Whytock *et al*., 2017).

In a survey of Scottish and England broadleaf woodland patches aged 10 to 160 years, older woodlands with mature trees were found to have higher total bird abundance and thus richness (Whytock et al., 2017). A study in Sweden (Felton *et al*., 2016a) found that in young oak production stands, generalist migrant bird species dominated, reflecting the simplified and homogenous forest structure. However, later in the rotation cycle, environmental conditions developed sufficiently to support a bird community composition more consistent with that associated with protected oak forests. The authors emphasize that these conditions are unlikely to arise if rotation lengths were to be shortened in these forests (Felton *et al*., 2016a). In a comparison of plant communities in plantations and semi-natural woodlands on the island of Ireland (Coote *et al*., 2012), one notable difference was the almost complete lack of understorey in plantations. Plantations are generally clearfelled at the point where it becomes profitable to do so, which is often before they develop the vertical structure that has been found to be particularly important to birds (Coote *et al*., 2012 and references therein). General impacts of shortened rotation times include reducing the availability of older trees and coarse woody debris and increasing the proportion of forest area in a clear-cut state (Felton *et al*., 2016). Each of these has the potential to impact on different elements of biodiversity.

A report for Forest Research (Saraev *et al*., 2017) found that although there is no universal biodiversity response to stand age, with variations between taxa and sites, overall there is more evidence of increasing species richness with stand age, than of a fall (or of no change). Since different taxa have different responses to increasing stand age, the overall biodiversity response can appear quite 'flat' as increasing and decreasing components cancel each other out (see also Saraev *et al.,* 2019). The usefulness of measuring overall species richness or abundance is therefore limited. Using this measure can furthermore favour early successional species at the expense of the woodland specialists most under threat. An individually targeted response, taking into account specific site, species and landscape characteristics, is required to meet particular conservation objectives in particular woodlands. However, the review concluded that the current state of knowledge and available methods generally do not permit the inclusion of biodiversity in optimum rotation length models at present (Saraev *et al.,* 2017). A further paper (Saraev *et al.,* 2019) assessed species richness in even-aged conifer stands across Great Britain ranging from 8- 10 to 70-250 years old. In most cases, no significant relationship was found between stand age and overall species richness (barring a few isolated exceptions). Some significant relationships were however found for specific taxonomic groups. The authors incorporated biodiversity values into current economic models of optimum rotation length. As might be expected, the strength of the effect depends on the value placed upon biodiversity relative to timber. If current published values are used, even in cases where there is a strong effect of stand age on biodiversity the calculated optimum rotation length was not always extended significantly. The authors note various caveats to this work: due to small sample sizes no deciduous forests or stands older than 70 years could be included in the analysis; mobile taxa such as mammals and birds were not considered; and only species richness was examined (not, for example, abundance or other measures of diversity) (Saraev *et al.,* 2019).

2.4 Thinning

Thinning is the selective removal of some trees to concentrate growth in a smaller number of better-quality individuals.

2.4.1 Soil carbon pools

Thinning can affect soil carbon fluxes in a variety of ways, for example by altering the microclimate and thus litter decomposition rates; by temporarily reducing litterfall; and by adding a pulse of organic matter to the soil in the form of brash left on site (Blanco, 2018; Clarke *et al*., 2015; Jandl *et al*., 2007). Thinning also increases the stability of the remaining trees. This is important for soil carbon, as uprooting of trees by wind-throw destroys soil structure, which makes protected carbon accessible to decomposers (Jandl *et al*., 2007). Belowground litter may increase in the years following thinning as root systems of removed trees die and decompose (Clarke *et al*., 2015). The effects of thinning are mostly seen in the litter and organic layers; less experimental evidence is available for the effect of thinning on the carbon pool in the mineral soil (Jandl *et al*., 2007). The net effect of thinning on the soil carbon of a particular site may be positive, negative or neutral (Blanco, 2018 and references therein; Clarke *et al*., 2015).

2.4.2 Above-ground carbon pools

In the immediate term, thinning reduces the carbon pool of the stand and increases the amount of carbon harvested (see 'Fate of harvested wood' section in the current report for a discussion of carbon storage in standing forests versus harvested products). Thinning is expected to have a negative impact on stand-level productivity initially because of the removal of photosynthetic material. However, the extent and duration of this effect is likely to depend on the intensity of thinning and how rapidly the growth and carbon uptake of the remaining trees can compensate (Saunders *et al.,* 2012). Year to year weather variations may also alter the response of a forest to thinning, by affecting rates of net stand carbon uptake and respiratory losses (Saunders *et al.,* 2012). The results of a study of a Sitka spruce plantation in Ireland indicate that thinning could have a negative, neutral or positive effect on carbon-draw down of a forest stand, depending on temperatures (Saunders *et al.,* 2012).

Thinning increases the stability of individual trees and thus reduces the risk of carbon losses due to disturbance (Jandl *et al.* 2007). Selective thinning has been found to reduce the adverse effects of climatic changes in some cases (Tang *et al.,* 2017).

Thinning (or wider initial spacing of trees) leads to faster growth rates of individual trees. In Sitka spruce there is a negative relationship between growth rate and wood density (Moore, 2011), which implies that an increase in growth rate might not equate with an equally accelerated growth in carbon stock. Körner (2017) argues that if trees grow faster, they will either arrive more rapidly at harvesting size or pass through their natural life span faster. When this takes take place in synchrony over large areas (as is the case for even-aged plantations), the resulting gain in carbon storage is transitory. Growth stimulation increases carbon turnover, but not the carbon residence time and therefore not the size of the carbon stock.

2.4.3 Biodiversity

Management prescriptions for biodiversity are more complex than for commercial timber production. However, thinning – even when the primary objective is increasing the commercial value of the timber produced – allows more sunlight to penetrate the canopy, which as a general principle can help support a wider range of species (CCC, 2018b). In broadleaved woodlands, absence of management leads to rapid closure of woodland canopy and increased shading, with a consequent loss of understorey vegetation and reduced woodland biodiversity, including breeding bird diversity and density (Burgess, 2014 and references therein). A study on the island of Ireland (Coote *et al*., 2012) concluded that the number of woodland species supported by even dense-canopied plantations can be enhanced through maintaining adequate below-canopy light levels by early and regular thinning. Other authors have similarly concluded that opening canopies, along with other measures such as leaving some deadwood in situ and retaining semi-natural patches and veteran trees, can be effective at maintaining biodiversity in plantations (Horák *et al.,* 2019 and references therein). One meta-analysis found that thinning had generally positive or neutral effects on diversity and abundance across all taxa (Egnell *et al*., 2016). However, the few European studies that examine effects of woodland thinning specifically on birds find little influence (Burgess, 2014). Temporal studies where gaps in canopy cover are created through management such as coppicing tend to show most change in the first 8 years post-management, before canopy closure occurs, so any benefits are likely to be short-term (Burgess, 2014 and references therein). A literature review of the impacts of forest practice in certified UK forests concluded that thinning has a positive or neutral effect on diversity of ground flora, with a paucity of studies on other taxa (Bellamy & Charman 2012).

2.5 Harvesting method

For forests where production of timber is a primary objective, the simplest approach is a "clearfell" management system in which a forest is created as a series of even aged stands which are clearfelled and then replanted or regenerated. The timing of clearfelling and regeneration of individual stands can be controlled to ensure maintenance of forest cover at the landscape level (Matthews *et al*., 2014). Under continuous cover forestry, a constant lower level of timber production is maintained while retaining canopy cover. There is also an important distinction to be made between traditional stem-only harvesting and whole tree harvesting, discussed below.

2.5.1 Soil carbon pools

Harvesting removes woody biomass that would otherwise have contributed to soil organic carbon stocks in the long term (Clarke *et al*., 2015), and has a disturbance effect on soils that can result in loss of carbon (Jandl *et al.,* 2007; Nave *et al.* 2010; Noormets *et al.,* 2015). The amount of soil carbon lost is partly site-specific and partly determined by harvesting methods used (as well as by the extent of site preparation for the next rotation) (Clarke *et al.*, 2015; Elofsson *et al.,* 2018; Nave *et al.,* 2010). Older studies estimate the amount of carbon lost from the organic layer of soil at up to 50% in the first 15 – 20 years after harvest, but more recent work has tended to reduce this estimate (Elofsson *et al.,* 2018; Clarke *et al.,* 2015 and references therein). The long-term net effect of harvest on soil carbon may depend on the extent of soil disturbance (Jandl *et al*., 2007). A global meta-analysis of temperate forests found that forest harvesting resulted in a small but significant reduction in soil carbon of 8% (Nave *et al*., 2010). This was entirely due to changes in the litter layer: no carbon impacts of harvest were detected in the mineral layer. Temporal trends suggested that the carbon loss from the litter layers, though substantial, were not permanent (Nave *et al*., 2010).

Clearfelling removes all aboveground tree biomass and disturbs the understorey layer. Litter inputs, and therefore carbon input to the soil, are in the short term drastically reduced or stopped (Blanco, 2018; Clarke *et al.,* 2015). However, in the medium term, harvest can stimulate the rapid growth of ground flora and young stands of trees that together could have a higher litter input compared to pre-harvest (Clarke *et al*., 2015). Harvesting with heavy machinery also disturbs the soil (Blanco, 2018; Clarke *et al.*, 2015). Soil mixing might increase decomposition of organic matter, or soil compaction might decrease it (Clarke *et al.*, 2015). After clearfelling, the soil is also exposed to more light, higher moisture content, and greater fluctuations in temperature, all of which can influence decomposition of both harvest residues and older soil organic matter (Clarke *et al*., 2015). Surveys of forest soils in Denmark 18 years apart revealed a trend of soil carbon loss following clearfelling and restocking (Callesen *et al.* 2015). Modelling of forest soils across Austria over the next hundred years indicated that increasing harvest rates would result in a slight decline in soil carbon stocks. Under a business-as-usual scenario, the carbon pool would increase over the same period (global warming of 2°C was assumed in both scenarios) (Hernández *et al.,* 2017). Some modelling studies have suggested that there could be long-term effects following clearfelling, with a 14% reduction in soil organic carbon after two 100-year rotations compared to pre-harvesting conditions (Clarke *et al.*, 2015).

Continuous cover forestry can have similar impacts on soil carbon to thinning (Jandl *et al*., 2007). Continuous cover forestry is considered to reduce soil carbon losses compared to clear-felling. However, following even selective harvest it can take decades for a new tree canopy to develop. During this time, carbon losses by decomposition can be higher than litterfall inputs, causing a net loss of carbon (Blanco, 2018 and references therein). Nevertheless, this effect is less than in clear-felling (Clarke *et al.,* 2015). Some studies have found that uneven-aged stands (as opposed to the even-aged stands created by clearfell systems) have soils that absorb carbon at a faster rate, possibly due to the maintenance of forest cover, reduced soil disturbance during harvest and/or niche complementarity allowing trees of different sizes to use resources more efficiently leading to higher productivity (Jonard *et al.,* 2017 and references therein). Continuous cover forestry is expected to lead to more stable carbon stocks in the litter layer but may not have any significant effects on the stabilisation of carbon in the mineral layers (Jandl *et al*., 2007). A modelling study of Norway spruce plantations in Austria found that a transition from clearfelling on a 90-year rotation to continuous cover forestry has considerable potential to increase carbon storage in the forest ecosystem as a whole (Seidl *et al.*, 2008). This benefit would be retained even at substantially lower average stocking levels.

When trees are felled, a varying proportion of the resulting material is harvested and removed from the forest. Stem-only harvesting can result in a pulse of organic matter being incorporated into the soil from decaying brash and harvest residues (Blanco, 2018, Jandl *et al.* 2007). This may explain the temporary increase in the carbon content of the soil that has been observed up to two decades after harvesting (Clarke *et al*., 2015). Conversely, decomposition of large amounts of needles and leaves will typically lead to increased soil respiration linked to a pulse of increased carbon dioxide release (Clarke *et al*., 2015; Noormets *et al.,* 2015). In the case of whole-tree harvesting, a much greater proportion of the biomass is removed from site (Blanco, 2018) and therefore smaller quantities of organic matter are returned to the soil (Thiffault *et al.,* 2011). A global review found no clear impact of whole tree harvesting on total soil carbon, though there was some evidence of negative effects on carbon stock in the litter layer (Thiffault *et al.,* 2011). In this review, whole tree harvesting appeared to have less of an impact on temperate forest soil carbon stock compared to boreal forests. A meta-analysis reviewed by Thiffault *et al.* (2011) found negligible differences associated with harvest method in hardwood and mixed forests, although significant positive effects of stem-only harvesting were observed in coniferous forests. The authors noted that none of the field studies reviewed extended beyond two decades, so conclusions about long-term impacts cannot be drawn (Thiffault *et al.,* 2011). Another review concluded that, on non-waterlogged mineral soils, a reduction in soil organic carbon content after whole-tree harvesting compared with stem-only harvesting may be likely (Clarke *et al*., 2015). Thiffault *et al.* (2011) suggested two factors that might explain the differing responses of forests to whole-tree harvesting. Firstly, they observed more negative impacts on carbon stocks in soils that were already poor in organic matter and/ or coarse-textured. Secondly, carbon from surface residues on sites in moderate and warmer climates is mainly respired as carbon dioxide and very little carbon is incorporated into the soil, while under wetter and cooler conditions much of the carbon in residues can eventually accumulate in the soil.

The impacts of biomass removal from a forest on productivity are highly site-specific (Egnell *et al.,* 2016; Whittaker *et al.,* 2011; Thiffault *et al*., 2011). The responses of a forest to the impacts of harvest will depend partly on the tree species present and their stage of growth (Thiffault *et al*., 2011). A global review of field studies found that soil phosphorus levels, and to a lesser extent nitrogen levels, are generally reduced after whole tree harvesting relative to stem-only harvesting. Whole tree harvesting is also expected to cause a greater drain on reserves of calcium, magnesium and potassium (Thiffault *et al.,* 2011). Where tree growth is limited by specific nutrients, removal of biomass can therefore lead to reduced productivity (unless fertilisers or ash are applied, which would also come with implications for greenhouse gas balance) (de Jong *et al.,* 2017; Clarke *et al.,* 2015). Removal of brash and stumps alters growing conditions for the next generation of tree seedlings, which could also affect productivity (de Jong *et al.,* 2017). Whole tree harvesting almost always improves initial tree seedling survival compared to stem-only harvesting because it creates more favourable growing conditions (Thiffault *et al*., 2011). Any limiting effects caused by nutrient depletion are generally seen in the regenerating stand several years after harvest, when growing trees are less influenced by microclimate and competition from accompanying vegetation, but have increasing nutrient requirements (Thiffault *et al*., 2011).

2.5.2 Above-ground carbon pools

Empirical modelling has shown that systems such as Continuous Cover Forestry, which contain greater age diversity, can provide larger long-term carbon stocks compared with single age stands (Sing *et al.,* 2017). One group of authors found that a multi-canopy structure allows new growth to quickly fill in the gap created by death of a large tree, meaning that such forests can continue to be net carbon sinks for centuries (Luyssaert *et al.,* 2008). Conversely, in single-age stands the carbon draw-down functions declines or even reverses with age (Luyssaert *et al.,* 2008).

Some evidence suggests that resilience to disturbance may be higher in multi-aged stands because of their greater ability to quickly recover to a pre-disturbance state (Tang *et al.,* 2017). A Forest Research note on climate change adaptation (Ray *et al.,* 2010) encourages forest managers to consider alternatives to clearfell systems. Continuous cover forestry is given as an example that may offer advantages for regeneration and establishment. Transforming single-aged stands to increase structural diversity does however increase the risk of wind disturbance with consequences for the loss of standing carbon stocks (Sing et al., 2017).

2.5.3 Biodiversity

Clearfelling a stand results in immediate loss of biodiversity prior to recolonization from nearby mature stands. Retaining mature trees within clearcut stands can mitigate this loss (Sing *et al.,* 2017). The habitat created by clearfell does, however, benefit some species including willow warbler and redpoll (Calladine *et al.,* 2015; Burgess *et al.,* 2015). A review of forestry practices in a UK context found that the early stages of replanted forest (following clearfelling) provide important habitats for many taxa, which are not found in the older forest stands (Bellamy & Charman, 2012). These replanted areas are also distinct from similar aged plantations in newly afforested plantations, since they retain some of the forest species accumulated during the first forest growth cycle as well as allowing colonisation by some open habitat species. This review concluded that, in general, clearfell and replant had a positive effect on biodiversity when compared to mature forest, in most situations resulting either in an increase in species or abundance after clearfelling (Bellamy & Charman, 2012).

Management intensity has been shown to influence species richness and abundance, with species that are dependent on the continuity of forest cover, deadwood and large trees negatively affected by more intensive management (Sing *et al*., 2017). Biodiversity within a forest tends to be greater in stands that are structurally diverse in terms of their age, species, patch edge, understorey and deadwood component (Sing *et al.,* 2017). A study in Sweden found that, relative to clearfelling, continuous cover forestry is expected to better maintain late successional forest conditions and associated micro-climates throughout the management cycle, specifically in relation to the continued availability of relatively mature trees and coarse woody debris (Felton *et al*., 2016). This would benefit many taxa. However, many naturally regenerating broadleaf tree species in these forests are pioneer species, which would potentially be adversely affected by continuous cover forestry approaches (Felton *et al*., 2016).

According to a review carried out for Forest Research (Saraev *et al.,* 2017), more evidence is needed on the impact of alternative forest management approaches such as continuous cover forestry and how these differ from traditional even-aged stands with respect to biodiversity. However, the same review did uncover certain characteristics of specific management systems that are beneficial or detrimental to aspects of biodiversity. These are given in table 2.

Large-scale removal of logging residues (as occurs in whole-tree harvesting) significantly decreases habitat availability for many deadwood associated species. The use of heavy machinery also increases the level of disturbance and may adversely affect soil and ground arthropod fauna and understorey vegetation (Felton *et al.,* 2016). Deadwood abundance is known to be important to some bird species including pied flycatcher and common redstart (Burgess, 2014). Meta-analysis of available data has shown that forest stands with less woody debris and standing deadwood can have significantly lower diversity and abundance of birds and invertebrates (Egnell *et al*., 2016). For many woodland invertebrates, birds and animals, structural attributes such as deadwood, dense low foliage and open space may be as important as the plant species present (Amar *et al.*, 2010). It may be possible to reduce negative impacts and maximise benefits for biodiversity by careful targeting of biomass harvest, avoiding intensifying management of forests that are highly biodiverse or contain threatened species, and leaving sufficient biomass of the type (for example coarse deadwood) required by the species of interest (de Jong *et al*., 2017).

Table 2: Forest management systems and biodiversity.

Adapted from Saraev *et al.*, 2017.

Silviculture	Characteristics identified as positive for biodiversity	Characteristics identified as detrimental to biodiversity
Clear cut	• Large (temporary) open spaces • Refuge for grassland species in intensively managed arable landscapes • Provision of edge habitat • Providing a diversity of habitats across the landscape	• Even-aged structure and generally high density • Lack of horizontal and vertical stand complexity • Structure favours generalists and excludes woodland specialists • Management technique precludes many species • Lack of natural regeneration • Lack of tree species diversity
Coppice	• Permanent and temporary open space • Standard trees • Varied ground flora • Structural diversity • Deadwood in abandoned coppice	• Lack of deadwood in active coppice • Lack of tree species diversity • Lack of structural diversity associated with abandoned or over mature coppice
Selection felling	• Stand continuity • Structural complexity • Standing biomass • Tree age distribution • Gap release and open areas • Horizontal diversity	• Few refuges for species susceptible to disturbance • Open areas can be too small to benefit a full suite of open habitat species • Absence of large veteran trees
Shelterwood	• Structural Diversity in mid storey • Canopy trees • Seedling regeneration	• Lack of open space, ground flora and microhabitats • Lack of horizontal diversity • Even aged structure and lack of mature forest

2.6 Conclusions

Society requires forests to provide a variety of ecosystem services, including timber production, carbon sequestration, biodiversity, water supply and recreation. It is not possible to maximise all of these at once, so management decisions will inevitably involve trade-offs. Some authors have asserted that prioritising the speed of carbon capture (as opposed to how fast carbon is lost from the system, how large the stock is when at equilibrium or for how long it is stored) is more likely to lead to trade-offs between carbon sequestration and the provision of other ecosystem services including biodiversity (Rodríguez-Loinaz *et al.,* 2013). They cite the transformation of natural forests into fast-growing plantations as an example of this. One review found that higher intensity forest management provides the greatest wood output but impacts negatively on biodiversity, health and recreation and water supply services (Sing *et al*., 2017). On the other hand, low intensity management/ no management is unsuitable for high production, but provides high levels of other services (Sing *et al*., 2017). Mapping approaches can help by revealing areas of conflict or areas of co-production of two or more ecosystem services, facilitating spatial prioritisation of management actions (Sing *et al.,* 2017). Forest managers may choose to spatially segregate areas where different ecosystem services are prioritised, or to attempt to integrate ecosystem service delivery with consistent management across the whole forest unit (Sing *et al.,* 2017).

The findings of the current literature review as they apply to UK forestry can briefly be summarised as follows. For fast carbon drawdown and high timber and biomass production, plant fast-growing conifers and harvest by clearfell on a short rotation (i.e. harvest soon after the trees reach their age of maximum growth). For large, long-term forest carbon stocks, plant slow-growing deciduous trees and manage under a low-intensity system such as continuous cover forestry or with a long rotation time (similar conclusions were drawn by Alonso *et al.,* 2012, in their evidence review for Natural England). Decide on thinning regime and harvest intensity (stem only versus whole-tree harvesting) on a case-by-case basis. For resilience to climate change and its effects (including increased pests and diseases), increase the diversity of tree species and ages within stands.

Given that different taxa benefit from different types of woodland management (see Table 2), any assessment of overall effects on biodiversity will inevitably involve a subjective choice between species assemblages (Burton *et al*., 2018). Most of the available evidence is based on impacts on single taxa at the plot scale. There is a need for more studies that are assess biodiversity more broadly at landscape scale and over the longer term (Burton *et al*., 2018). However, based on the findings of the current report, the following broad recommendations would seem to hold true. Protect old-growth semi-natural woodlands. Favour native broadleaved species for new plantations. Manage plantations (including by diversifying the species planted) so as to increase light levels and structural diversity and provide sufficient undisturbed and deadwood habitat.

3 Forestry on deep peat

3.1 Context

Active peatlands absorb carbon dioxide through the growth of peat-forming plants such as Sphagnum species. When left undisturbed, the carbon absorbed in this way is stored within the peat for millennia. Peatlands also emit carbon dioxide and methane. Nitrous oxide can also be important, especially when peatlands are exposed to nitrogen in fertilizer as is the case in some peatland afforestation, or are affected by long range diffuse nitrogen pollution from agricultural sources. Methane and nitrous oxide have global warming potentials 34 and 298 times, respectively, greater than carbon dioxide over a 100-year period when climate-carbon feedbacks are considered, and thus can be important contributors to global warming (IPCC 2013). Methane, however, has a much shorter lifespan in the atmosphere than carbon dioxide (Artz *et al.*, 2013). Taking all these figures into account, it is generally accepted that healthy peatlands have a net cooling effect on the atmosphere in the long term (Artz *et al.*, 2013; Bain *et al.,* 2011; Evans *et al.,* 2017). They are also a large long-term carbon store: estimates indicate the UK's blanket and raised bogs currently store at least 3.2 billion tonnes of carbon (Bain *et al*., 2011).

The UK's peatlands are estimated to occupy a total area of around 3.0 million hectares (Evans *et al.,* 2017). However, only an estimated 22% of this area remains in a near-natural condition, with a further 41% under some form of semi-natural vegetation but affected by human activity. Arable cropland occupies 7% of the UK's peat area, grasslands 8% and woodland 16% (Evans *et al*., 2017). Overall, the UK's peatlands are estimated to be emitting a total of approximately 23,100 kt CO2e yr^{-1} of greenhouse gases (Evans *et al*., 2017).

Drives towards drainage and afforestation in the second half of the twentieth century and particularly the 1980s have resulted in a legacy of conifer plantations on deep peat in the UK (Sloan *et al*., 2018). About 10% of UK blanket bog has been planted with commercial forestry (Bain *et al*., 2011). Afforestation of deep peat involves extensive drainage and other site preparation activities such as fencing to exclude deer, building of roads and deep drains alongside them, construction of quarries and borrow pits to surface the roads and fertilising of young trees. Lowering the water table of peatlands generally results in increased emissions of carbon dioxide and decreased emissions of methane (Sloan *et al.,* 2018). There may also be changes to carbon fluxes through aquatic pathways via dissolved and particulate organic carbon. The potential loss of carbon from peat soils on initial afforestation (involving drainage and ploughing) has been estimated at 20 – 25% of the total carbon in the peat (studies reported in Perks *et al.*, 2011). The disturbance caused by harvesting and replanting also releases a large pulse of carbon (Artz *et al*., 2013).

Forestry plantations on deep peat have proved to be less productive than had been expected (Andersen *et al*. 2016, as cited in Hermans, 2018); Payne *et al.*, 2018a), and are now widely considered to have detrimental impacts on ecosystem services that outweigh economic benefits. The UK Forestry Standard (Forestry Commission, 2017) has a guideline presumption against establishing new forests on soils with peat more than 50cm in depth and on sites that would compromise the hydrology of adjacent bog or wetland habitats, although woodland creation on deep peat soils that have historically been highly modified may be allowed in some circumstances. This guideline does not consider possible impacts on biodiversity of adjacent bog or wetland sites (for example through invasive conifer seedlings or edge effects on ground-nesting birds) beyond the requirement to assess potential impacts on priority habitats and species that applies to all new forest and woodland proposals. Forest managers are also required to “consider the balance of benefits for carbon and other ecosystem services” before restocking existing plantations on deep peat (Forestry Commission, 2017).

Many of the UK’s plantations on deep peat are reaching economic maturity and are due to be felled. According to a Forest Research report on peat soils in Scotland (Morison *et al.,* 2010), the greenhouse gas implications of restocking previously planted and disturbed deep peat sites are different from those of new planting on peat soils which currently support non-woodland vegetation. For each plantation, a decision has to be taken on whether to restock the plantation or to restore the peatland (Sloan *et al.,* 2018). Where biodiversity conservation is an objective, there are clear benefits to peatland restoration (Sloan *et al.,* 2018; Bain *et al.,* 2011; Payne *et al.,* 2018). The case for greenhouse gas emissions has until recently been less clear-cut (Sloan *et al.,* 2018), but, as discussed in the following sections, this is an active area of research and the carbon arguments for restoring afforested bog are becoming stronger.

3.2 Biodiversity considerations of peatland restoration

Peatlands contain a large number of unique species, albeit many of them at low abundance (Payne *et al.,* 2018a). Initial afforestation leads to significant – in some areas, total – loss of this biodiversity (see for example Stroud *et al.,* 1987). Planting trees leads to fundamental changes to the ecosystem, with drying of the peat, loss of carbon, shading from the tree canopy and nutrient addition. Afforested sites typically include a greater abundance of generalist and woodland species and far fewer peatland specialists (Avery & Leslie, 1990). The loss of Sphagnum species from afforested sites is particularly significant given these species’ role in peat formation (Payne *et al.,* 2018a).

Afforestation also impacts on biodiversity on open peatlands adjacent to the forest plots (Wilson *et al*., 2014; Payne *et al.,* 2018a; IUCN, 2010). For example, evidence from the Flow Country and beyond suggests that woodland plantations in otherwise open landscapes increase predation pressure on ground-nesting birds (Mark Hancock, RSPB, personal communication July 2019). Dunlin and golden plover occur at lower densities near plantations, with this ‘edge effect’ extending hundreds of metres beyond the plantation itself (Wilson et al., 2014). In northern Scotland, removal of commercial conifer plantations for blanket bog restoration could benefit waders breeding on surrounding open ground through reduced edge effects, as well as eventually allowing populations to re-establish on the restored area itself (study in review, cited in Douglas *et al*., 2014; Wilson *et al.,* 2014).

In restoration projects, recovery of peatland biodiversity may be slowed by forestry legacy, such as the release of nutrients from brash and litter years after the trees have been removed (Payne *et al.,* 2018a). In some sites, certain non-target species can become dominant during restoration and may inhibit the recovery of typical bog species. However, there is good evidence that restored former forestry sites on deep peat can make substantial progress towards recovery in 10-20 years, including the return of moisture-loving vegetation (Hancock *et al.* 2018). Experiments have been made to speed vegetation recovery through translocation of plants and application of micropropagated plant products in an effort to restore cover of typical species, particularly Sphagnum mosses (Payne *et al.,* 2018a).

3.3 Climate impacts of conifer plantations on deep peat

3.3.1 Carbon loss from peat

Evidence points to ongoing loss of carbon from peat soils associated with forestry plantations (Alm *et al*. 2007; Laine *et al*. 2009; Simola *et al*. 2012; von Arnold *et al*. 2005). Long-term records from an afforested deep peat site in Scotland show an average reduction of 56.8 cm (or 13 %) in the depth of peat under forest stands 50 years after afforestation (subsidence indicates but does not directly demonstrate carbon loss as peat compaction is also probable). Most of the subsidence was related to the initial drainage and planting and there was little change after the initial loss. However, a reduction in peat depth was also seen throughout the open areas between forestry plots, even though these areas have not been directly drained, and it appears that in some areas this loss may continue over the life of the plantation (Sloan *et al.* 2019). As plantation trees grow and close canopy they can draw down the water table beyond the level initially achieved through drainage (Anderson & Peace, 2017). The extent of the horizontal impact zone of plantations on the hydrology of adjacent peat has yet to be determined (Sloan *et al.,* 2018). Hermans (2018) found that peat decomposition rate under 30-year old conifer plantations in the Flow Country was around 126.8 g C m^{-2} y^{-1} (or 1.27 t C ha^{-1} y^{-1}, comparable with the 1 t C ha^{-1} y^{-1} calculated by Hargreaves *et al.,* 2013). Lab experiments showed that peat from the forest plantation decomposes faster than the peat from the near pristine blanket bog, even under the same temperature and water level. This is probably due to changes in biochemical composition of peat leading to different microbial communities. The actual difference in the field conditions accentuates this difference, i.e. peat is decomposing faster in the forest than the bog both because field conditions under trees are conducive to faster decomposition and because the microbial communities in the peat under plantations promote faster decomposition at any given temperature and water level (Hermans, 2018).

3.3.2 Carbon uptake by trees

Trees planted on deep peat (like any trees) draw down carbon from the atmosphere and store it. Some studies (discussed below) have found that in some circumstances the rate of this carbon draw-down can exceed the rate of carbon loss from the peat. However, it is important to understand that the residence time of carbon in plantation trees and harvested wood products is much lower than in peat in an intact bog. Payne *et al*. (2018), in their evidence synthesis for the Valuing Nature Programme, found that trees on drained and afforested bogs are likely to have higher rates of carbon fixation than natural bog vegetation. According to a report for ClimateXChange (Vanguelova *et al.*, 2018), the majority of studies show that afforested drained peats are likely to act as net carbon sinks while the trees are growing, despite large peat losses – although they state that more data is needed, a point echoed by Payne *et al.* (2018), among others. In Hermans' study (2018) at age 30 years the annual carbon fixation by the plantation trees is enough to more than cancel out annual carbon losses from the peat, so the plantations are a net sink at this particular point in the rotation (this result does not consider the large carbon loss during initial afforestation). Two earlier reports to government indicated that afforestation of peatlands in Scotland resulted in an initial loss of soil carbon for approximately four years, after which the forests became significant carbon sinks (Harrison *et al.* 1997; Jones *et al.* 2000). Similarly, Hargreaves *et al.* (2003) looked at net ecosystem carbon dioxide exchange on afforested deep peat sites in Scotland. They found that the freshly drained and ploughed peatland is a carbon source, but that four to eight years after planting the peatland becomes a sink, initially due to carbon fixation by recolonized ground vegetation but thereafter due to the growing forest. They concluded that forestry had a net carbon benefit over the lifespan of a plantation, but pointed out that the carbon stock in most peatlands greatly exceeds the stock that can be added by growing trees, so eventually peatland restoration would become beneficial (Hargreaves *et al*., 2003). This study has been criticised by Lindsay *et al.* (2010, as reported in Hermans, 2018), who pointed out that the "undisturbed" site used as a control in this study is in fact itself highly modified and has a lower carbon accumulation rate than truly pristine sites, and that the oldest plantation in the study is only 26 years old (whereas according to these authors harvest usually takes place at 60 years, although note that 35 – 45 years is the typical rotation length that is usually quoted, e.g. see Moore 2011). Extrapolating the peat loss results up to 60 years would result in no net carbon benefit from forestry. In Ireland, modelling indicates that carbon dioxide loss from drained and planted peatland is comparable to the uptake of carbon by the forest (Byrne *et al.* 2005). One study of afforested boreal peatland found that fertilisation (with wood ash or nitrogen/phosphorus/potassium fertiliser) enhanced the greenhouse gas sink function of the plantation by increasing tree growth. However, over timescales longer than a few decades, fertilisation had a net warming effect on the climate due to increased soil emissions (even before emissions from fertiliser manufacture and transport are considered) (Ojanen *et al.,* 2019).

3.3.3 Overall greenhouse gas balance on afforested peatlands

Fully assessing the greenhouse gas balance requires models which account for methane and nitrous oxide in addition to carbon dioxide (as noted by Vanguelova *et al.,* 2018). Hermans (2018) finds that afforested areas in the Flow Country are a weak sink for nitrous oxide, and that methane fluxes are lower in currently afforested sites than restored sites. Studies from countries outside the UK similarly indicate that drainage of peatlands for afforestation results in a decrease in methane, and an increase in nitrous oxide emissions (Alm *et al*. 2007; Haddaway *et al*. 2014; Laine *et al*. 2009; Maljanen *et al*. 2010; Minkkinen *et al*. 2002). In terms of 100-year global warming potential, the increase in nitrous oxide emissions may equate to the decrease in methane emissions, suggesting little overall net effect (Haddaway *et al*. 2014). Minkkinen and Laine (2006, as cited in Hermans, 2018) estimated that the waterlogged ditches in a forest emit at least as much methane as is consumed by the rest of the forest. This would mean that most drained afforested peatlands are small sources of methane. One study finds that peatland forests may act as either significant soil carbon sinks or sources (Ojanen *et al*. 2014), and another two suggested that afforested peatlands actually act as carbon sinks (Maljanen *et al*. 2010; Minkkinen *et al*. 2002). However, there was a high degree of uncertainty in measurements of these gas fluxes in several of these studies, and a common conclusion was that improved measurement precision over a longer time period and spatial scale is required.

In addition, it should be noted that differences between peatland types and forestry methods mean that any comparison of afforestation of peatlands between Fennoscandia (where most of the above studies originate) and the UK must be made with extreme caution (IUCN UK Peatland Programme, 2014). Fennoscandian peatland forestry (e.g. see Haapalehto *et al.* 2011) usually involves draining an existing natural woodland on wet peat to improve native tree growth and production, with drains spaced about every 10-20m. The situation in the UK and Ireland (where open blanket bog with no natural woodland is ploughed and drained much more intensively and non-native conifers are planted and fertilised) is very different.

3.3.4 Loss of carbon in aquatic forms

Carbon is also lost from afforested peatlands in aquatic forms. Disruption caused by activity such as ploughing, tree planting and drain creation and maintenance (particularly the large drains that serve forestry roads) is associated with increased concentrations of dissolved and particulate organic carbon in streams draining the forest stand. Later, disruption to the peat surface caused by tree thinning or felling can lead to further aquatic carbon loss for several years after the trees are removed (Sloan *et al.,* 2018).

3.3.5 Summary

In summary, the literature indicates that after the initial rapid loss caused by draining and ploughing there is an ongoing, if slower, loss of carbon from afforested peat. Offsetting this loss there can be gains of carbon stored in tree biomass, litter and new soil organic matter. It is worth mentioning that there are carbon costs associated with establishing and maintaining forestry on peatlands, in particular fossil fuel use in ploughing, planting, fencing, fertilising, drain maintenance and road building. These climate impacts have not been well quantified (Sloan *et al.,* 2018). Restoration activities also have carbon costs in the short term, but in the long term restoration means that roads, drains and fences no longer have to be maintained, whereas these would be ongoing costs if forestry rotations were continued. The net greenhouse gas balance of an afforested site may be determined by the specific management practices employed (IUCN UK Peatland Programme, 2014). The carbon absorbed by the growing trees may exceed the carbon lost by the peat at certain points in the rotation or averaged over a whole rotation, so that the plantation can be described as a carbon sink at least until the trees are harvested. However, few studies consider all greenhouse gas fluxes or explicitly compare the carbon storage potential of harvested wood products with that of restored peat. There is a generally-agreed need for more data. In particular, Vanguelova *et al.* (2018) identify a clear need for long term studies using different planting ages (chronosequence studies) to ensure robust results when evaluating the impacts of afforestation and restocking on soil carbon stocks, as short-term impact studies are likely to provide misleading conclusions. Chronosequence studies do, however, have limitations, in particular the difficulty of identifying suitable sites for comparison given that forestry practices have changed very significantly over the last 40 years.

3.4 *Climate impacts of restoring peatlands*

Restoration of afforested peat involves a number of activities, such as felling, drainage blocking and rewetting and a range of other developing restoration techniques such as re-profiling. These affect the hydrology, soil temperature, vegetation and evapotranspiration of the system, with complex implications for overall greenhouse gas balance (Yamulki *et al.,* 2013). The timescales for restoration, the degree to which it is successful, and changes in greenhouse gas fluxes as restoration proceeds, are not well understood (Hambley, 2019; Hermans, 2018). Because greenhouse gas fluxes are naturally highly variable in different years, it is difficult to attribute changes to restoration activities in the short term (Hermans, 2018).

3.4.1 *Methane fluxes*

The impacts of restoration on methane fluxes are poorly understood, but research generally points to increased methane emissions post restoration (Hambley, 2019). Whether the overall effect is beneficial for climate mitigation therefore depends on whether the increase in production of methane is smaller (in terms of carbon dioxide equivalent radiative forcing) than the decrease in production of carbon dioxide. The Global Warming Potential of methane[1] is 62 for a period of 20 years, 23 for a period of 100 years, and 7 for a period of 500 years (IPCC, 2001). Values of CO_2e are often calculated using a 100-year time horizon. Government policy, however, tends to focus on shorter time frames (for example, the UK's commitment to 'net zero' emissions by 2050 (Priestly, 2019)), raising the open question of what is the most relevant timeframe over which to measure the climate effects of restoration. As pointed out by Artz *et al*. (2013a) in their research summary for ClimateXChange, using longer time horizons results in reduced emphasis on the impact of methane.

3.4.2 *Evolution of greenhouse gas budgets during restoration*

Carbon budgets of peatland sites undergoing restoration, especially in the early stages, are different to those of pristine bogs and are a function of the condition of the peatland prior to restoration, the hydrological conditions achieved through restoration and the stage of restoration (Artz *et al.*, 2013). Strong growth of recolonising vegetation and slow rates of decomposition (due to the time taken for the microbial community to adapt to the new conditions) can mean recently restored peatlands are stronger net sinks of carbon dioxide than mature bogs (Artz *et al.*, 2013; Hambley *et al.,* 2019). One study at Forsinard Flows, using modelling based on satellite data, found that peatland sites undergoing restoration reach the carbon assimilation potential (Gross Primary Productivity) of near-natural bog sites 5-10 years after restoration commences (Lees *et al.,* 2019). This is one measure of successful restoration, but the authors emphasise that consideration should also be given to changes in respiration, other carbon fluxes, water table, and vegetation communities (Lees *et al.,* 2019). Increasing the long-term carbon store depends on re-establishing peat

[1] Global Warming Potential compares the impact of emissions of a given greenhouse gas over a specified timeframe with the impact of the same amount of carbon dioxide emissions. For example, the GWP for methane over a time frame of 100 years is 23. This means that emissions of 1 million metric tonnes of methane is equivalent to emissions of 23 million metric tonnes of carbon dioxide. The carbon dioxide equivalent (CO_2eq) for a gas is derived by multiplying the tonnes of the gas by the associated GWP, so 1 tonne of methane would be expressed as 23 tonnes CO_2eq . Source: https://ec.europa.eu/eurostat/statistics-explained/index.php/Glossary:Carbon_dioxide_equivalent

forming species, which may have lower carbon dioxide uptake rates than the initial colonisers (reference within Morison, 2012). Morison (2012), in a report for ClimateXChange, states that methane emission rates can also be high initially as the presence and cover of species which can enhance methane emissions increase on re-wetting, and subsequently emissions may decline over decades (Morison, 2012).

Two recent studies in Forsinard Flows (Hambley *et al.* 2019; Hermans, 2018), using different greenhouse gas measurement techniques, agree that by 16 years, peatlands undergoing restoration are net sinks in terms of carbon dioxide exchange and total greenhouse gas exchange – although still weaker sinks than the near-pristine bog control site. Hambley *et al.* (2019) examined net ecosystem carbon dioxide exchange at two formerly afforested sites which had undergone felling and interceptor drain blocking (no furrow blocking) 10 and 16 years prior to the study. The 10-year restoration site was a net source of carbon dioxide (80 g m^{-2} yr^{-1}, expressed as CO2-C), but the 16-year site was a net sink (although only -71 g m^{-2} yr^{-1}, compared to -114 g m^{-2} yr^{-1} on the near-pristine control site). The author estimated the switchover time from a net carbon dioxide source to a net sink at around 13 years post-restoration (note that this study did not look at methane or dissolved or particulate organic carbon). Restoration was far from complete at this point: the vegetation on the restoration sites still differed significantly from undamaged bog and the water tables remained well below the original peat surface (Hambley *et al.* 2019). Hermans (2018) measured peat decomposition and fluxes of carbon dioxide, methane and nitrous oxide along a chronosequence of sites that had undergone felling and drain blocking 0 – 17 years ago. Peat decomposition was fastest under plantations and slowed down as a result of restoration. However, overall carbon dioxide respiration was unchanged over the chronosequence, probably due to increasing respiration from the recolonising bog vegetation balancing out the decreasing respiration from the peat. All sites were a weak sink for nitrous oxide. Methane fluxes increased with restoration age, as predicted. Nevertheless, the site that had undergone restoration 17 years ago was found to be a net greenhouse gas sink (-130g CO_2e m^{-2} y^{-1}), although weaker than the near-pristine bog (-307.80g CO_2e m^{-2} y^{-1}) (Hermans, 2018).

The Scottish Government is carrying out a 5-year project, due to complete in 2021, which focuses on the Forsinard Flows sites plus an eroded blanket bog site in the Cairngorms. A mid-programme update outlines the findings so far (Artz *et al.,* 2019). Restoration sites show a partial recovery of hydrology indicated by consistent elevated water table, but full recovery had not been achieved more than 20 years following restoration management. Monitoring of carbon dioxide fluxes indicates that the restored former plantation sites are returning to a more natural state in terms of their carbon dioxide exchange budgets. Data is not yet available for methane fluxes (Artz *et al*., 2019).

3.4.3 Aquatic forms of carbon

The carbon balance of peatlands is also influenced by fluxes of dissolved and particulate organic carbon (DOC and POC) that leave the site in watercourses. DOC may degrade and enter the atmosphere downstream, usually as rapid emission of carbon dioxide. The fate of POC is less certain (Sloan *et al.,* 2018). Muller *et al.* (2015) studied short-term changes to the hydrochemistry of water on deep peat sites in the Flow Country when plantations were felled, the trees mulched and the drains blocked. Dissolved organic carbon (DOC) in the water on-site increased noticeably when restoration activities commenced, although very little was actually transported to the nearby stream. The authors noted that decomposing biomass from the mulched trees was a very significant source of DOC. Two years after the trees were felled, DOC concentrations on-site were relatively steady but still much higher compared to the intact bog site. The authors anticipate that DOC will continue to drop up to 20 years post-felling, as bog vegetation re-establishes. A similar study in 2017 at Forsinard Flows (Gaffney, 2017) also found significant increases in DOC in pore and surface water on-site in the first year following restoration. Like Muller and colleagues (2015), Gaffney found no significant impacts of restoration on exports of dissolved carbon from the site. He hypothesised that as more restoration is carried out within the catchments and the proportion of plantations felled increases, greater impacts on streams and rivers may be observed. Vinjili (2012), studying a river catchment with a combination of intact bogs, plantations and sites undergoing restoration in the Flow Country, concluded that restoration activities had no significant impact on DOC release. Instead, climatic changes related to precipitation and temperature, coupled with water yield capacity of the sub-catchments, were identified as the main driver of DOC fluxes. 57 - 95% of the DOC export occurred during 5 - 10% of the high flows, i.e. storm events. From these studies, it therefore appears that although restoration activities may increase release of DOC on-site (as indeed may harvesting and restocking), there is little impact on the amount of carbon exported from the site in water.

3.4.4 Vegetation recovery

The re-establishment of peatland vegetation is a key aspect of restoration, from a greenhouse gas balance point of view as well as for biodiversity objectives (Gaffney, 2017; Hermans, 2018; Worrall *et al.,* 2011; Hancock *et al.,* 2018). Plants of certain genera, such as Sphagnum and Eriophorum, are key to the recovery of a bog's peat-forming function (Hancock *et al.* 2018). Even partial restoration can restore some sink function and, crucially, protect the remaining carbon store in the peat (as noted by Vanguelova *et al.,* 2012 in their report for Forest Research). However, several reports emphasize that bog vegetation takes time to recover once rewetting has taken place (Hermans, 2018; Vanguelova *et al.,* 2012; Hancock *et al.,* 2018; Anderson & Peace, 2017; Artz *et al.,* 2019). Gaffney (2017) found elevated levels of phosphorus, potassium and ammonium in pore and surface water following forest-to-bog restoration. The author concluded that more than 20 years is required for complete recovery of water chemistry to bog conditions, which may help explain why some of the characteristic bog plants do not come back readily following rewetting. Some authors have found that the rate of recolonization of bog vegetation depends strongly on the stage of the conifer crop at removal: faster restoration can occur in non-canopy-closed bog forests where some of the previous bog vegetation still exists (Hancock *et al.,* 2018). According to a Forest Research report (Vanguelova *et al.,* 2012), where remnants of natural peatland vegetation survive in the wetter and less disturbed area of plantations (e.g. ditches, rides), this speeds up recolonisation by providing sites from which Sphagnum can spread once the trees have been removed. Hancock *et al.* (2018) compared vegetation on restoration sites, intact bog sites and afforested peat sites in the Flow Country. They found that in the six years after restoration began, vegetation developed towards bog-like conditions. In the subsequent eight years, overall vegetation change stalled, and spatial variability increased. Specifically, drier areas such as plough ridges and steeper slopes showed poor restoration progress. This study also found that the restoration plots differed most strikingly from bog controls by their greater cover of grasses (*Poaceae*). These grasses are linked to poorer restoration outcomes, and their abundance may reflect legacies from drainage prior to afforestation, or nutrient enrichment (Hancock *et al*., 2018). By contrast, a ten-year experiment in Scotland (Anderson & Peace, 2017) found that vegetation on ridges and the original surface are responding to restoration in the right successional direction (albeit slowly), whereas plough furrows are succeeding towards a different plant community from that found on unplanted bog. The authors suggest that this may be because the furrows are deeper and steeper-sided than the natural small depressions found on undisturbed bog, with higher shade levels and nutrient concentrations playing a part (Anderson & Peace, 2017).

Ongoing management of bogs undergoing restoration to remove colonising tree seedlings may be necessary (Anderson & Peace, 2017), especially where damage to the peat has changed the hydrological function of the landscape (Vanguelova *et al.* 2012; Artz *et al.,* 2018). Anderson & Peace (2017) suggest that further research is needed to determine whether timing and method of felling can be optimised to reduce conifer regeneration.

3.4.5 Restoration techniques

The specific methods employed for restoration have an impact on the likely timescales and outcomes. It has been shown that tree felling is necessary to achieve peatland recovery (Anderson & Peace, 2017). Drain blocking (at least of interceptor drains) is also beneficial (Anderson & Peace, 2017), although in flatter areas simply ceasing maintenance of drains can allow vegetation to colonise and block them (Hancock *et al.,* 2018). In early restoration projects of the 1990s and early 2000s it was common for main drains to be blocked but plough furrows to remain unblocked, which may have slowed down the rate at which the water table has recovered (Hambley, 2019; Gaffney, 2017), especially on sloping ground (Hancock *et al.*, 2018). Leaving some drainage in place can also result in spatially heterogeneous vegetation recovery as drier patches remain within the restoration area (Hancock *et al.,* 2018). If the water table remains low, organic matter such as dead tree roots left in the soil after felling will decompose and release carbon dioxide (Hermans, 2018). Forest Research authors writing in 2012 noted some evidence that where peat cracking is at an advanced stage (as may be the case in older plantation stands or second rotations), restoration of the water table can be challenging (Vanguelova *et al.,* 2012). However, restoration techniques are being improved and new ones developed that should help to overcome these challenges (Gaffney, 2017).

An important element of restoration is whether trees are felled to waste (i.e. left in situ, with or without mulching) or harvested, and if the latter, how much of the brash is removed. Where trees are small and poorly-formed (as was the case for many early restoration projects) they were generally felled to waste, but as plantations have matured more recent restoration work has necessitated harvesting and extracting the trees in an increasingly wide variety of ways (Neil Cowie, RSPB, personal communication July 2019). Organic material left on the surface is a source of dissolved organic carbon and nutrients, which may impede the recovery of bog vegetation (Gaffney, 2017; Hancock *et al.,* 2018). Conversely, operations to remove material from the felled plantation constitute extra disturbance to the peat, which may itself have consequences for greenhouse gas emissions and bog recovery (Gaffney, 2017). A ten-year restoration experiment in Caithness, Scotland found that it is not necessary to remove felled trees, with bog vegetation able to develop even if they are left on the ground (Anderson & Peace, 2017). One study of a Scottish restoration site where trees were felled to waste calculated that it would take the restored bog between 15-73 years to accumulate the same amount of carbon that would be lost during decomposition of the felled trees (reported in Vanguelova *et al.,* 2012). New methods of restoration are being trialled which may help accelerate and enhance the recovery of restored peatlands (Gaffney, 2017). Studies in Scotland indicate that restoration techniques like additional brash crushing and furrow-blocking raise the water table depth significantly higher than the original fell-to-waste technique and have a positive effect on ecological recovery (Artz *et al*., 2019). Hancock *et al.* (2018) make the general recommendation that restoration management should take place in parallel with research and monitoring, within an adaptive management framework.

3.4.6 Impacts of climate change on peatland restoration

The impacts of climate change on current peatlands, their carbon balance, and on the prospects for restoration of afforested peatlands, are not certain. Modelling suggests a long-term decline in the distribution of actively growing blanket peat, especially under high emissions scenarios, although it is emphasised that existing peatlands may persist for decades or even more or less indefinitely under a changing climate (reported in Vanguelova *et al*. 2012). Intact bogs are expected to be more resilient to the predicted changes than damaged bogs (research reported in Vanguelova *et al.* 2012). Evidence suggests that one of the most important thresholds for successful restoration to functioning peatland is the re-establishment of appropriate hydrological conditions. The likely impacts of climate change on the functioning of restored and natural peatlands at particular sites are not fully understood (see for example Artz *et al.,* 2019).

3.4.7 Recommendations arising from research to date

Since 1990, an estimated 95,000 ha of UK peatland have been subject to some form of active restoration intervention (Evans *et al.,* 2017). However, the majority of restoration activities have taken place in areas of modified blanket bog, which produce modest emissions sources per unit area. Restoration activities to date are estimated to have generated an emissions reduction since 1990 of 423 kt CO_2e yr^{-1}, but restoring sites that produce higher emissions per unit area, including plantation forest, could provide much greater emissions abatement (Evans *et al.,* 2017).

Morison (2012) states that the net greenhouse gas balance change for specific restored sites is determined by the change in water table depth and the fertility of the site. He suggests that restoration should be targeted at sites with poorest tree growth and with most potential for successful and early restoration of peatland to a net carbon sink. He further recommends that the net greenhouse gas balance of afforested peatland restoration will be improved if harvested wood products are used in industry, displacing more high-carbon materials such as concrete, and if tree removal occurs close to maturity or normal rotation length. However, in practice trees from peatland plantations are rarely of sufficiently high quality to use for timber (Sloan *et al.,* 2018), and high windthrow risk means that many plantations are being harvested before the usual rotation length (Mark Hancock, RSPB, personal communication July 2019).

The findings of the recent PhD research on Forsinard Flows (Gaffney, 2017; Hambley, 2019; Hermans, 2018) have been integrated in a policy briefing for ClimateXChange (Hermans *et al.* 2019). This concludes that restoration sites older than 15 years help to combat climate change by storing more greenhouse gases than they emit, taking into account carbon dioxide, methane and nitrous oxide. In the short term, disturbance associated with restoration activities tends to increase greenhouse gas emissions, but this is compensated by the amount of net climate cooling after 15-20 years. The results confirm the benefits of forest removal on deep peats where conifer yields have been low. In addition to habitat improvements, the long-term climate benefit of restored peat bogs is unlikely to be matched by forestry. Newer management techniques, such as intensive drain and plough-furrow damming, may help faster recovery of carbon sequestration (Hermans *et al.* 2019).

3.4.8 Summary

In summary, the impacts of restoration activities on greenhouse gas fluxes are complex and depend on, among other things, the conditions of the specific site prior to restoration and the particular restoration methods used. Recent restoration projects have provided valuable data. The emerging picture is that the ongoing sink function of bogs can be restored over a period of years, even before the bog vegetation makes a full recovery. New restoration methods are being developed and trialled, which could improve the timescales and extent of recovery.

3.5 Conclusions

It was stated at the start of the current section that the greenhouse gas case for restoring peatlands is still being built. Given the timescales over which emissions targets apply, the option to restock plantations may appear attractive. However, it is vital to understand the fundamental difference between peat and commercial forestry plantations as a carbon store. Plantations are felled after 35–45 years or so (Moore, 2011) and their carbon is released back into the atmosphere at a rate depending on the uses of the wood and the decay rate of the material left on site (Artz *et al.*, 2013). Forestry crops on UK peatlands are often of such poor quality that much of the wood goes for pulp, fuel and other short-lived uses (Sloan *et al.,* 2018) (see also 'Fate of Harvested Wood' section of the current report). Even if plantation landscapes are managed so as to maintain the pool of carbon in standing wood, there is a 'saturation' point after which no additional net sequestration takes place (Hargreaves *et al.*, 2003). By contrast, healthy peatlands store carbon and continue to absorb more from the atmosphere over periods of millennia (Sloan *et al.,* 2018; Bain *et al.,* 2011). A decision to restock a plantation must be seen in this light: any net gain in greenhouse gas balance over the coming decades may be at the expense of a much larger loss over the long term. The longer a plantation is left standing, the more carbon will be lost from the peat, and the harder it may be to eventually restore a functioning bog. A loss of only 5% of UK peatland carbon would equate to the total annual UK anthropogenic greenhouse gas emissions (Bain *et al.,* 2011). The fact that current understanding of the greenhouse gas fluxes of intact bog, plantations on deep peat, and restored peatlands is incomplete would seem to argue for a precautionary approach: protect the existing carbon store. In the long-term, intact peatland is a more secure carbon store than timber (as noted by Payne *et al.,* 2018 in their Valuing Nature report), as well as being home to unique communities of plants, birds and other taxa.

4 Forestry on shallow peat

4.1 Climate impacts of forestry on shallow peat

Soils with an organic layer of 30 – 50cm are termed shallow peat or organo-mineral soils. According to an evidence synthesis for the Valuing Nature Programme (Payne *et al*., 2018), differences in greenhouse gas budgets between deep and shallow peat are unknown. The current assumption in policy is that forestry on shallow peat does not lead to any ongoing change in soil carbon stocks. There is some evidence to support this, although it seems far from conclusive. A report produced for ClimateXChange in 2018 reviewed the evidence (peer-reviewed and grey literature) on the greenhouse gas impacts of forestry on peaty soils (Vanguelova *et al*., 2018). It concluded that carbon is lost from the peat layers over the first 30 years of afforestation due to the disturbance by soil preparation for afforestation, clear-felling and reforestation, but beyond the first rotation this could be compensated for by carbon accumulation in the litter and upper organic soil horizons. Furthermore, there may be reductions in methane emissions if drainage has sufficiently lowered the water table (Vanguelova *et al*., 2018). A study in north England aimed to quantify the impact of afforestation with Sitka spruce on carbon stocks of peaty gley soils (Vanguelova *et al*., 2019). The change of soil carbon stock from the previous land use (rough grazing of heather moorland/ blanket bog) and the relationship with forest age was studied using a chronosequence. The study found that the overall effect on soil carbon stocks was neutral over the time span of two forest rotations, as the loss of carbon from peat layers (-0.35 t C ha-1 y-1) was compensated by the carbon accumulation in surface organic layers (0.73 t C ha-1y-1).

If it is the case that there is no ongoing carbon loss from the soil, the net impact of forestry could be assumed to be positive (net carbon draw-down from the atmosphere) because of the carbon accumulation in the trees. One study found that soil carbon was 48% lower under a Sitka plantation on shallow peat than in a nearby unplanted area. However, soil carbon recovered to pre-planting levels by second rotation and the plantation acted as a carbon sink once standing stock was included (Zerva *et al*., 2005). An illustrative scenario of Sitka spruce afforestation on organo-mineral soil is presented by Matthews *et al*. (2014). It was generated using the CARBINE forest carbon accounting model. On afforestation, carbon stocks initially decrease due to losses from soil. After about 16 – 17 years, the net balance reaches zero due to accumulation of carbon stocks in trees, and thereafter is positive (i.e. net draw-down of carbon from atmosphere). The carbon stock in the soil recovers its initial levels at around year 40. Note: this scenario does not appear to include any harvesting (Matthews *et al*., 2014). Similarly, a modelling exercise using Forestry Commission carbon look-up tables and other generic published figures estimated that conifer plantations on shallow peat soils in Eskdalemuir generate net carbon benefits of 7.3 tonnes carbon dioxide equivalent per hectare per annum, averaged over 100 years (Greig, 2015). In this report for Confor the assumption was made that afforestation has no long-term impact on the soil carbon store. Given the large absolute size of this store, this assumption has a significant impact on the model's conclusions. Furthermore, a significant proportion (nearly 70%) of the assumed carbon benefit was associated with harvested wood products (as a carbon store and as a substitute for construction materials and fossil fuel energy). This sort of calculation necessarily involves a large number of assumptions (see 'Fate of Harvested Wood' in the current report). The remainder of the benefit was carbon stored in tree biomass. Vanguelova *et al.* (2018) in their literature review for ClimateXChange conclude that it is "very probable" that moderate and high productivity forests on shallow peat soils with limited disturbance provide a substantial net soil carbon uptake over the forest cycle. Generally, there are net losses of soil carbon in the first forestry rotation, balanced out by gains in the second and subsequent rotations. This is because uptake of carbon dioxide by the forest, and its subsequent transfer into the soil, is greater than losses from soil decomposition (Vanguelova *et al.,* 2018).

A number of studies have explored the impacts of different management practices on carbon balance of afforested shallow peatlands. Several studies were carried out in an afforested area on peaty gley soils at Harwood, Northumberland. Clearfelling resulted in a decrease in the amount of soil carbon stored (Zerva *et al*. 2005) but a fall in carbon dioxide emissions for 10 months post-clearfelling compared to a 40 year old forest (Zerva & Mencuccini 2005). Ball *et al*. (2007) found that annual emissions of carbon dioxide were lower in the 20 year old forest than in either the 30 year old forest or clearfelled sites. Emissions of methane were significantly increased by clearfelling in all three studies which measured this greenhouse gas (Ball 2007; Zerva *et al*. 2005; Zerva & Mencuccini 2005). However, while nitrous oxide emissions increased with clearfelling in two of these studies (Zerva *et al*. 2005; Zerva & Mencuccini 2005), they were highest in the 30 year old forest in the other (Ball 2007).

A further study at Harwood investigated the effects on methane emissions of different site preparation methods: drainage, mounding and fertilisation (Mojeremane *et al.,* 2010). Mounding consists of mechanically excavating peat to a depth of 30–40 cm and heaping it upside down next to the pit. The results of the study indicated that water table depth was the major factor determining methane fluxes. Drainage decreased soil moisture and increased temperature, both of which influenced microbial activities causing methane production to decrease. Carbon dioxide emissions were however higher in the drained plots in both years of the study. Mounding significantly increased methane fluxes, especially from the hollows between the mounds. There are several mechanisms that could contribute to this result: for example, the pools of stagnant water created provide substrate for methanogenic bacteria; mounding buries the litter and organic layers beneath the mineral layers; and the machinery used causes soil compaction which may reduce methane oxidation. Fertilisation significantly increased methane fluxes in the first year, probably due to effects on the microbial community (Mojeremane *et al*., 2010).

According to research carried out for the Scottish Executive (Scottish Executive, 2007), harvesting leads to carbon loss from increased decomposition rates (as well as the direct removal of carbon in harvested biomass) which is then offset by uptake of carbon in the re-growing vegetation. The net balance is likely to be site-specific and will be determined to a great extent by harvest residue management and site environmental conditions (Scottish Executive, 2007). A study in Kielder Forest, northern England, assessed the impacts of whole tree harvesting (WTH) and fertilisation (compared with stem-only harvesting) on a peaty gley forest soil under second rotation Sitka spruce, 28 years after harvesting (Vanguelova *et al.,* 2010). Plots subject to WTH had higher organic carbon content and total nitrogen content, apparently because retention of residues under stem-only harvesting increases the rate of mineralisation of soil organic carbon and nitrogen stocks. Fertilisation similarly increased mineralisation rates and thus reduced soil carbon and nitrogen content. However, the magnitude and direction of such effects will be site specific, with the greatest changes in soil carbon and nitrogen expected in soils with deep organic layers and high initial stocks. Soil moisture was found to be significantly higher under WTH because of poorer tree growth leading to less rainfall interception and evapotranspiration. The authors suggest that in the longer term, WTH (as opposed to stem-only harvesting) on highly organic soils could be beneficial both for soil carbon storage and sequestration and peat layer protection (Vanguelova *et al.,* 2010). An evidence review for ClimateXChange found that stump removal, on the other hand, results in high loss of carbon from both shallow and deep peat soils (Vanguelova *et al*., 2018).

In summary, shallow peat soils are likely to be vulnerable to carbon losses during the tree establishment phase and through erosion losses during ground preparation. A second period of vulnerability may be associated with forest harvesting both through physical disturbance and accelerated leaching losses of dissolved organic carbon (Scottish Executive, 2007). Because of soil heterogeneity, vertical gradients in organic matter content and bulk density it is problematic to draw general conclusions from field studies on shallow peat. The extent to which results from other systems, for example peatlands and lowland agricultural soils can be extended to upland organo-mineral soils is also very uncertain (Scottish Executive, 2007). Future models need to take account of the different stabilisation mechanisms of organic carbon in both the peat and the mineral layers (Vanguelova *et al.*, 2018). In the context of shallow peat soils in the UK uplands, the evidence from direct measurements is inconclusive about effects on soil organic carbon stocks following a change in land use from semi-natural grassland / moorland to forestry (Scottish Executive, 2007). There is evidence that organic matter accumulates at the soil surface, but the long-term fate of this material is uncertain. The extent of carbon stabilisation is likely to depend on the tree species, soil type, site nutrient status, site hydrology and climate. There is some evidence for loss of older carbon deeper in the soil profile (Scottish Executive, 2007). Other authors, however, have found evidence that carbon released from the peat layer through afforestation can move downwards and be sequestered in the mineral soil underneath (as reported in Vanguelova *et al.,* 2018). This effect could be significant where the mineral soil has high clay content (Vanguelova *et al*., 2018).

The overriding conclusion from the above brief review seems to be that the net effects of forestry on shallow peat soils are highly variable. It may not be appropriate to apply general models to specific sites based on the broad definition of "shallow peat"

4.2 Biodiversity

Some shallow peats are associated with priority habitats such as wet heath and EU Annex 1 habitat, which support (or could be restored to support) important breeding populations of priority birds such as curlew. Shallow peats often occur adjacent to or within a mosaic of deep peat and are hydrologically linked. Therefore, planting on shallow peat could also impact the hydrology, vegetation assemblages or priority species on nearby deep peat. However, when the above exclusions are met, planting native woodland on shallow peat may in some cases benefit biodiversity. For example, establishing native carr woodland around the edge of intact or restored raised bogs can attract woodland birds, insects and other invertebrates. Dominated by alder, sallow and birch, the damp shady conditions provided by carr woodland also benefits ferns, mosses, liverworts and lichens (Broads Authority, 2017).

5 Nature-based forestry

5.1 Context

This section briefly reviews forest management approaches that have the primary aim of benefitting biodiversity and examines their carbon benefits.

Götmark (2013) suggests four habitat management alternatives for temperate forests where conservation is the main aim:

(1) Minimal intervention, allowing continued succession and natural disturbances. This allows the development of old-growth forests, which are rare in many regions and benefit many taxa.

(2) Traditional management. Such approaches, which include coppicing in the UK, favour species associated with past cultural landscapes. Many such species are now threatened (red-listed) due to lack of suitable habitat, and may need active management. Traditional management can be used to increase heterogeneity of forest types and to favour species that require non-closed canopies.

(3) Non-traditional management can be used to produce old-growth characteristics (for example thinning the canopy to promote the growth of large individual trees) or specific forest composition, or to favour one or a few tree species that benefit biodiversity. This is a flexible approach that can be targeted towards specific conservation aims, for example restoring native woodland from plantation.

(4) Species management may be applied where a highly-valued species with well-known habitat requirements is threatened with extinction, or where a keystone species can help improve the ecological function of a forest. Götmark considers rewilding to be a form of species management, as it emphasizes introduction and the regulatory role of a few large mammalian predators, and the role of large herbivores in large tracts of 'wild' land.

The following sections broadly follow the above structure. In addition, I briefly consider the publicly-funded creation of new forests on farmland.

5.2 Minimum intervention

Paillet *et al.* (2010) carried out a meta-analysis to compare species richness between unmanaged and managed forests in Europe ('unmanaged' here meaning not influenced by direct human disturbance for at least 20 years). They found that species richness tended to be higher in unmanaged than in managed forests, but the response varied widely among taxonomic groups. Saproxylic beetles, bryophytes, lichens and fungi suffer from the reduction of micro-habitat availability and diversity in managed forests and showed higher species richness in unmanaged forests. Conversely, the species richness of vascular plants, many of which are favoured by frequent disturbances such as canopy opening, tended to be higher in managed forests. Forests had to remain unmanaged for at least 20 years before they become more species-rich than managed forests. The authors infer that after management stops, the dynamics of the ecosystem gradually restore appropriate conditions for the recolonisation of species that are dependent on typical unmanaged forest substrates. This recolonisation also depends on the regional species pool and the dispersal ability of species, and can take more than 40 years for some taxa. The authors state that their results provide arguments for the conservation of unmanaged forests (Paillet *et al.,* 2010).

In the UK context, semi-natural forests have been subject to management for centuries, and their native wildlife communities reflect this. Therefore, appropriately tailored management is often necessary to conserve species of conservation interest. The condition of many semi-natural woods in England is threatened by neglect or inappropriate management (as concluded by Alonso *et al.,* 2012 in their evidence review for Natural England). Based on monitoring of Sites of Special Scientific Interest, it has been concluded that the most serious and widespread reasons for unfavourable ecological condition of priority woodland habitats are excessive deer browsing, uniform structure, non-native species and uncontrolled grazing by livestock (Forestry Commission, 2010). One review examined habitat and structural data collected in British woodlands in the 1980s and again in 2003–2004 (Amar *et al*., 2010). The majority of woods were high forest (as opposed to coppice), as is now the case for the majority of woodland in Britain as a whole. Between the 1980s and 2003-2005, the sub-canopy was found to increase in nearly all locations. An increase in ash at the expense of oak was noted. Standing dead trees and dead- wood on the ground both increased, with the increase in the number of standing dead trees suggesting some self-thinning of canopy trees (Amar *et al*., 2010).

There are significant areas of broadleaf forest in England which are currently subject to minimal management. These were mainly planted from the 1940s onwards. Sometimes these woodlands were subject to management in early years, but active management has now been abandoned. This has resulted in dense stands of trees with closed canopies and little understorey vegetation. There are policies aimed at bringing them back into management to meet both biodiversity and carbon objectives (Matthews *et al*., 2014, CCC, 2019). The extent to which biodiversity does in fact benefit from a resumption of management will be determined by the type of management that is adopted. Table 2 summarises the characteristics of different forest management systems that are found to be positive or negative for biodiversity. There are pros and cons to each system, but clearfelling in particular precludes the survival of many species and favours generalist species over woodland specialists.

5.2.1 Carbon

Forests containing deadwood and trees that are grown into mature and old growth phases have been shown to store more carbon (Sing *et al.,* 2017), and even very old unmanaged forests can be large net annual sinks (Jandl *et al*., 2007; Luyssaert *et al.,* 2008). It would be expected that the absence of soil disturbance from harvesting or other operations would facilitate the build-up of a large, long term carbon store in the soil (see for example Jandl *et al.,* 2007). Estimates of long-term carbon stocks are generally greater in stands and forests where no harvesting occurs (Geng *et al.*, 2017).

Evidence for carbon draw-down in old-growth stands in the UK is limited, but international evidence clearly shows the importance of old-growth stands (Burton *et al.,* 2018). Authors of a recent comment in Nature (Lewis *et al*., 2019) find that, because of the regular release of carbon from plantations during harvesting, natural forests are forty times better than plantations at storing carbon (global average). A global review of growth rates of individual trees found that, contrary to common assumption, rates of carbon gain for most tree species increases continuously with tree mass (Stephenson, 2014). The authors suggest that the rapid growth of large trees indicates that, relative to their numbers, they could play a disproportionately important role in forest effects on global carbon cycle. The results of a global review of temperate and boreal forests challenged the assumption that old-growth forests are at carbon equilibrium (Luyssaert *et al.,* 2008). Forests as much as 800 years old can continue to draw down carbon. These old forests tend to contain a small number of large trees (because of competition and self-thinning in unmanaged forests, or thinning in the case of managed forests). In such forests, when a large individual tree dies, new growth (of understorey vegetation or the sub-canopy) accelerates. The carbon accumulation as a result of this growth takes place much faster than the carbon loss from decay of the dead tree, so the system as a whole continues to draw down carbon. In this way, old stands with high densities and a multilayer canopy structure can continue to maintain biomass accumulation for centuries (Luyssaert *et al.,* 2008). Körner (2017) reasons that faster rates of tree growth simply mean trees reach harvesting size/ natural age of death more quickly, and so do not lead to an increase in the size of the carbon pool over the long term. For a forest over the long term, the growth and decay of trees can be considered to balance out. The carbon pool can only be increased by increasing the residence time of carbon in the forest, through a demographic shift towards older trees (Körner, 2017). Moomaw *et al.* (2019) find that leaving forests 'intact' and allowing them to grow to their ecological potential – a practice they term 'proforestation' – is a more effective climate change mitigation strategy than afforestation or reforestation. They point out that afforestation requires a large amount of additional land, and newly-planted stands of trees take time to become significant carbon sinks, whereas 'proforestation' has large immediate benefits for climate change as well as other ecosystem services. The work is carried out in a United States context but the conclusions are more widely applicable (Moomaw *et al.,* 2019).

Modelling by Matthews *et al*. (2014) suggests that ceasing management of conifer plantations could lead to carbon draw-down of about 5 tC ha^{-1} yr^{-1} over the first 20 years, dropping to 2 tC ha^{-1} yr^{-1} for a time horizon of 100 years. The equivalent figure for a broadleaf forest is a constant 2 tC ha^{-1} yr^{-1} for the first 100 years (although this rate of draw-down would drop off eventually). As mentioned elsewhere in the current report, it should be noted that a reduction in UK wood production might result in increased imports of wood from abroad, resulting in no global reduction in felling.

As mentioned above, there is interest in bringing UK broadleaf woodlands that are currently subject to minimal management back into active management. The Committee on Climate Change (CCC, 2018b) recommends that efforts should be increased to deliver the policy commitment to bring 67% of England's forests back under active management (from 59% currently), and to seek to extend the ambition where the evidence supports this. The stated intention of this approach is to provide carbon benefits by increasing the annual carbon draw-down rate while providing a source of timber and fuelwood to substitute for fossil fuels and high-carbon construction materials (but see 'Fate of Harvested Wood' in the current report for discussion of the latter) (Matthews *et al*., 2014, CCC, 2019). However, the evidence discussed in the preceding paragraphs (Sing *et al.,* 2017; Jandl *et al*., 2007;

Luyssaert *et al.,* 2008; Körner, 2017; Moomaw, 2019) suggests that unharvested, old-growth forests can continue to draw down carbon as well as store it in large amounts over long periods. It must also be noted that resuming harvesting in currently unmanaged woodland would result in a one-off decrease in in-situ carbon stocks, with associated carbon emissions (Matthews *et al*., 2014; Geng *et al.,* 2017). In a report to DECC, Forest Research acknowledge that the carbon impacts of bringing woodlands back into management are uncertain and that further research is needed to identify optimal silvicultural regimes specifically in support of restoring management in so-called 'neglected' forests (Matthews *et al*., 2014).

5.3 Coppicing

Coppicing is a way of producing a continuous supply of small-diameter wood for fuel and other uses. In the past it was widely practiced in the UK, but since the 1940s there has been a shift to high forest alongside the general decrease in active management of broadleaved woodlands (Amar *et al.,* 2010; Burgess, 2014). However, coppicing is still prevalent across parts of Europe (Lee *et al.*, 2018). In the UK, interest in restoring woodland biodiversity through reinstating traditional woodland management is growing. This may involve resuming active coppice management, or alternatives such as singling (the removal of all but one stem from former coppice stools), which aims to create a more varied age and vertical structure throughout the forest (Burgess, 2014).

Several taxonomic groups, including species of high conservation value, require both living tree and deadwood habitats, and semi-open habitats or canopy gaps. Coppicing (and other traditional extensive management such as pasturing) can create and maintain this mix of conditions (see also Table 2). The decline of these practices, alongside intensified deer browsing in lowland England, is leading to the conversion of coppices to high-forests, the closure of canopies and the loss of understorey structures, all of which may threaten many highly specialised taxa (Burrascano *et al*., 2016; Amar *et al.,* 2010).

A study in southwest England explored the impacts on hole-nesting birds of management aimed at restoring abandoned oak coppice to a more natural and varied vertical structure (Burgess, 2014). The results indicated that, despite this intervention being implemented specifically to benefit woodland birds, it had very little influence on their populations over 55 years. However, this was probably because the management applied was insufficient to make a substantial difference to the openness and vertical structure of managed plots, with the oak canopy cover remaining continuous (Burgess, 2014).

5.3.1 Carbon

Coppiced woodlands have different characteristics to high forests. They are found to be more resistant to water stress because of reduced competition between trees and because stools have deeper roots than seedlings, but have reduced stand fertility and smaller carbon stocks (Lee *et al*., 2018). Coppiced shoots tend to have a higher vigour than young plants (Ray *et al*., 2010, Forestry Commission research note), which should benefit the carbon draw-down rate. In a comparison of active and abandoned oak coppice in Turkey, soil carbon was found to be 116.0 - 140.3 tonnes C ha^{-1} (coppiced) and 128.1–236.2 tonnes C ha^{-1} (abandoned) (see Table 1) (Lee *et al.,* 2018).

If coppicing involves removing significant volumes of timber from a woodland, it reduces the carbon stock of that woodland (CCC, 2018b). This is an example of a trade-off between biodiversity and carbon objectives. A possible approach is to carry out management as necessary for biodiversity objectives, but leave the felled stems to rot in situ (or stack them if this is considered necessary for other objectives, e.g. amenity). This would have the added biodiversity advantage of providing more deadwood habitat. This approach would initially enhance carbon stocks on site (as the remaining trees increase their growth rates) but the deadwood would eventually decay and release its carbon (Matthews *et al*., 2014), so in the longer term the carbon loss would be the same as for harvesting the wood.

5.4 Converting plantations to native woodland

During the mid to late 20th century, many sites that once supported ancient semi-natural woodland were converted to plantation forests to provide a source of timber. Such sites are known as plantations on ancient woodland sites (PAWS). There are 200,000 ha of PAWS in Britain, mainly under non-native conifers but with a significant proportion under plantations of broadleaves (Harmer *et al*., 2013). Restoration aims to restore their biodiversity value, and focuses on safeguarding the survival of remnant features such as broadleaved trees, woodland ground flora species with slow rates of dispersal and veteran trees. This generally involves removing the introduced species of trees, either gradually or as a one-off felling, depending on specific site conditions and objectives. Other native species may also be present and benefit from restoration including invertebrates occurring in deadwood and leaf litter, and the micro-organisms associated with undisturbed ancient woodland soils (as noted in Harmer *et al*., 2013).

Barsoum *et al.* (2016) produced a review for the Forestry Commission of research on converting planted non-native conifer to native woodlands. The following paragraph summarises their findings on biodiversity from the peer-reviewed literature. Large, single species conifer stands, especially those with a dense canopy, tend to support mainly mosses, lichens and vascular plant species such as ferns that tolerate acidic substrates and low light and nutrient conditions. Native woodland species abundance and diversity can increase substantially following conversion from conifer to native broadleaved, although for some species colonisation may take a long time or might never happen. Even partial conversion to more open canopy tree species (e.g. Scots pine and many broadleaved species) has been shown to result in changes to the ground vegetation community, although not necessarily a return to typical native woodland flora. Where native woodlands do not occur in close proximity and light levels have not been maintained for the duration of a non-native conifer crop on a site, the developing ground flora may comprise mostly ruderal and moorland species depending on adjacent and pre-afforestation land use. More abundant understorey layers in response to an opening up of the canopy are likely to increase the nesting opportunities for woodland birds and reduce risks of predation. Forest specialist bat species also favour high levels of structural heterogeneity and have been shown to benefit following the introduction of a native broadleaved component in managed conifer stands. Soil macro and microfauna have also been observed to respond positively to the addition of broadleaved species to or conversion of conifer stands.

5.4.1 Carbon

The process of conversion is likely to involve a short-term loss of carbon because of removal of biomass (felling unwanted trees) and disturbance to the soil. The extent and duration of this effect will depend on the conversion method used, e.g. clearfelling the plantation or gradually removing trees to maintain a canopy while native species recolonise, and on the fate of the carbon in the harvested timber.

Replacing conifers with broadleaved trees will affect the carbon balance of the woodland. In general, broadleaved forests have a slower carbon draw-down rate but larger long-term carbon stocks than coniferous forests (see the section entitled "Tree species choice in commercial forestry" in the current report). After conversion, the forest will most likely be subject to less intensive management than the original plantation. This is likely to further increase the forest carbon stock (see "harvesting method" and "minimal intervention" sections within the current report.

5.5 Rewilding

Rewilding is an approach that is attracting increasing interest in the UK. It involves allowing natural ecosystem processes to re-establish in the landscape, and in some cases (re)introducing key species where they are absent (Rewilding Britain, *online*). In many cases this can eventually result in new woodland becoming established.

The impacts on different components of biodiversity will depend on the habitats present initially and once rewilding has commenced. For example, the conversion of semi-natural grasslands to forest (whether by abandonment leading to succession or through active afforestation) often leads to a strong decline in plant species richness (Burrascano *et al.*, 2016).

Some have argued that natural regeneration is more beneficial for biodiversity than active afforestation, especially in sites close to existing woodland, because it promotes the establishment of appropriate species and genotypes and produces a more 'natural' woodland in terms of structure and composition (Walker, 2003; references within Barsoum *et al.,* 2016). The Monk's Wood Wilderness is an experimental site in Cambridgeshire. Following at least a century of continuous cultivation, it was abandoned in 1961 and left to follow natural succession. The site is surrounded by ancient woodland on three sides. Botanical surveys in 1998 and 2003 demonstrated that the site had reverted to an oak-ash woodland and was gradually becoming more similar to the surrounding ancient woodland, although significant differences remained. These differences were attributed to seed availability and dispersal mode (Walker, 2003). This site is considered an example of relatively rapid woodland regeneration. Given the poor dispersal and colonisation abilities of many woodland species, the fact that many woodland plant species do not remain dormant in the seedbank for long, and the isolation of the majority of new woodland sites from available propagule sources, natural regeneration may take a long time to produce anything resembling semi-natural woodland in most cases (Walker, 2003; references within Barsoum *et al.*, 2016). On sites where natural regeneration takes place, early successional native tree species such as birch and willow are typically the first colonisers and can dominate stands for many decades (references within Barsoum *et al.,* 2016). These early successional stages themselves provide valuable habitat for some bird species such as willow tit and marsh tit.

5.5.1 Carbon

As for biodiversity, the carbon implications of rewilding depend on the start and 'end' habitats. There is evidence that rewilding arable land can lead to significant carbon benefits. A major rewilding project is taking place on the Knepp estate, West Sussex. Until 2001 the land was subject to intensive arable and pastoral agriculture. Since this point, conventional farming has ceased and the vision is for herds of livestock to graze across the majority of the Estate to allow the vegetation to establish as it may have done in the past under the influence of large herbivores. In a report to Defra (Hodder *et al*., 2010), scientists modelled the carbon implications of this project. They concluded that, if the vegetation develops as projected, the carbon stock of the land will increase by 55.1%. This is due to the expected increases in area of neutral grassland and broadleaved woodland at the expense of arable land. A study in Russia and Kazakhstan examined changes to soil organic carbon in the top 20 cm of soil following abandonment of large areas of arable land (Kurganova *et al*., 2015). Succession of natural vegetation resulted in an increase in soil organic carbon stocks across all regions. Initial rates of carbon accumulation were very high, possibly because the land was very carbon-depleted at the point it was abandoned. The rate of carbon accumulation declined over 20 – 50 years but soil organic carbon did not reach its new equilibrium until 60 – 80 years or more after the cessation of arable farming. The establishment of permanent vegetation cover on former arable soils, as well as contributing to soil carbon stocks, provided protection to the soil (Kurganova *et al*., 2015).

On the other hand, the conversion of semi-natural grasslands to forest (whether by abandonment leading to succession or through active afforestation) may not result in any significant overall gain in carbon stock (Burrascano *et al*., 2016 – see also 'Development of Forest Soils' in the current report).

5.6 Small-scale afforestation of arable land

Each of the governments of the UK provides funding to support private afforestation through their Rural Development Programmes. Stated objectives include biodiversity, water protection and carbon sequestration[2]. Evaluations of such woodlands have detected moderate benefits for taxa associated with open and young woodland habitats. However, these woodlands vary widely in terms of local character and landscape setting and the relative importance of these attributes in determining their biodiversity value is largely unknown. It also remains unclear how species assemblages will develop as these woodlands age (Humphrey *et al.,* 2015 and references therein).

Some studies of small-scale woodlands in arable landscapes have been carried out. Whytock *et al.* (2017) found that broadleaf woodlands planted on previously agricultural land provide

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[2] England https://www.gov.uk/government/publications/countryside-stewardship-woodland-creation-grant-manual-2018
Northern Ireland https://www.daera-ni.gov.uk/sites/default/files/publications/daera/Forestry%20Grant%20Scheme%20Information%20Booklet.pdf
Scotland https://www.ruralpayments.org/publicsite/futures/topics/all-schemes/forestry-grant-scheme/ and Wales https://gov.wales/glastir-woodland-creation.

highly favourable habitat for generalist woodland birds. Based on their study, they speculatively suggest that the expansion of broadleaf woodland cover in the UK during the past 30 years may have contributed to population increases or stability for some woodland bird species. Woodland characteristics such as stand structure, habitat heterogeneity and tree species composition are clearly important in determining biodiversity value (Humphrey *et al*., 2015; Fuller *et al.,* 2017). Birds appear to benefit from larger patches of woodland (Whytock *et al.*, 2017; Bellamy *et al*., 2004), with one study concluding that as a simple rule of thumb, patches larger than 5 ha should be created when the aim is to benefit generalist woodland bird communities, although much larger woodlands (more than 30 ha) may be required to benefit woodland specialists (Whytock *et al.*, 2017). Landscape characteristics, including amount of woodland in the area and connectivity between habitat patches, are also important for various taxa (Humphrey *et al*., 2015; Bellamy *et al*., 2004).

5.6.1 Carbon

Woodland creation on arable soils will generally increase soil carbon stocks as well as above-ground carbon over the medium term (see the section entitled "Development of forest soils" in the current report). The long-term carbon balance of the woodland created depends upon factors including species planted and how they are managed (see elsewhere in this report).

5.7 Conclusions

An estimated 18% of the UK forest area is managed for conservation of biodiversity (BEIS, 2016). There are few studies and no consistent method for assessing the effect of afforestation on more than one ecosystem service at a time, for example on biodiversity and carbon sequestration. This is a significant unmet need for evidence-based policy making (Burton *et al.,* 2018).

Based on this short review, it appears that in general managing forests for biodiversity generates more synergies with maximising the long-term carbon store, and more trade-offs with maximising the rate of carbon draw-down. A similar conclusion was drawn by Rodríguez-Loinaz *et al.* (2013). An exception is the practice of removing biomass to open up the canopy to benefit biodiversity, e.g. through coppicing.

Although there is a lack of empirical evidence for the effects of different biodiversity components on forest carbon balance, it is clear that the identity, relative abundance, number, and spatial arrangement of species are all likely to have an impact on carbon sequestration (Díaz *et al*., 2009). Biodiverse woodlands can generally be expected to be more resilient to threats (including climate change) than low-diversity plantations (Ray *et al*. 2010, Forestry Commission research note). A theoretical additional benefit of storing carbon in biodiverse forests is that such forests tend to hold more cultural value and policy importance than plantations, which could improve the chances of the forest and its carbon stores being preserved in the long-term (Díaz *et al*., 2009).

6 Fate of harvested wood

Note: for brevity, "bioenergy" in this section refers solely to forestry-based bioenergy.

The forest management decisions described in previous sections, as well as affecting the carbon balance of the forest itself, influence the products that are obtained from the forest. The uses of this harvested wood have a key impact on the overall greenhouse gas balance of the forestry system (Matthews *et al*., 2014; CCC, 2018b).

The biodiversity implications of different forestry approaches are not discussed in this section as they are covered elsewhere in the report. The focus of the current section (and the report as a whole) is on the UK: there are carbon and biodiversity issues for international forestry which are different to the UK and which are not covered here. It is also important to note that this section is focused on scientific knowledge and does not attempt to assess the policies governing the use of HWP and bioenergy. The actual climate implications will depend strongly on standards and policies (for example on forest management practices, carbon accounting and offsetting), and to what extent they are enforced. Finally, the likelihood and implications of future technological advancements, in particular carbon capture and storage, are not explored. It is worth noting that the climate change strategy proposed by the Committee on Climate Change (CCC, 2018b) depends heavily on the existence of such technology in the near future.

6.1 Background

The carbon sequestered in woody biomass is ultimately released back into the atmosphere. Trees or parts of trees that are not harvested will eventually die and decay in situ. Some of the carbon released from decaying wood enters longer-term carbon storage in the forest soil (Egnell *et al.,* 2016; CCC, 2018b). Disturbance events, such as fire, disease or clearfelling, may cause a large release of stored carbon from the various components of a forest ecosystem (Matthews *et al*., 2014). If wood is harvested, it may be burned, which immediately releases the carbon into the atmosphere. Alternatively, it may be made into harvested wood products (HWP) which may be long or short-lived, and may be reused and recycled, but which will ultimately decay and release their carbon into the atmosphere in the form of carbon dioxide or methane (Geng *et al.*, 2017).

Numerous authors (for example Geng *et al*., 2017; Noormets *et al*., 2015; Ter-Mikaelian *et al.,* 2015) emphasise that to gain an accurate picture, HWP and bioenergy life-cycle carbon analysis needs to be integrated with forest carbon balance analysis, whereas many previous studies have focused on one or the other of these. A similar point is made by Valatin (2008), who reviews several carbon accounting approaches and find that they either cover carbon substitution benefits of harvested wood products or carbon balance of the forest being harvested, but not both.

6.2 The no-harvest baseline

When calculating the climate impacts of harvesting for bioenergy or HWP, a critical consideration is what would have happened to the biomass otherwise (Ter-Mikaelian *et al*., 2015). One possibility is that the trees would have been left standing, i.e. a 'no-harvest' baseline scenario. Some authors have concluded that the no-harvest scenario is preferable to harvesting for bioenergy or HWP, i.e. that greater climate change mitigation can be achieved by leaving the forest standing and thus increasing its carbon stores. Münnich Vass *et al.* (2016), on the basis of empirical modelling of European forests, find that it would be economically optimal to manage forests to maximise sequestration, with the production of HWP reduced and bioenergy completely phased out. The authors urge caution in interpreting these results due to limitations of the study (for example, the study only considers longer rotations as a forest management measure and excludes other options such as thinning and afforestation of new land which could increase net sequestration). This study only considered standing biomass, not litter or soil. Another modelling exercise (involving one of the same authors) was recently carried out (Elofsson *et al.,* 2018). The authors model carbon pools in forest biomass, soil (the organic layer plus the top 20cm of the underlying mineral layer) and HWP, as well as fossil fuel consumption. They also conclude that the most cost-effective way for the EU to meet its carbon targets would be to successively reduce harvests in order to maximise ongoing sequestration in standing forests. Authors of a study of coppiced oak stands in Turkey conclude that overall net sequestration would be greater if coppicing ceased, even taking into account displacement effects when the coppiced material was used for energy (Lee *et al.,* 2018). Similarly, modelling of temperate hardwood forests in the north-eastern US states finds that, even when the carbon sequestered in HWP is taken into account, unmanaged forests sequester 39 – 118% more carbon than any of the active management options evaluated (Nunery *et al.* 2010). Moomaw *et al.* (2019) promote a policy of 'proforestation' (leaving forests ecologically intact), and state that the climate case for harvesting wood is based on questionable assumptions. They point out that, in practice, inefficient logging practices result in substantial instant carbon release to the atmosphere, and only a small fraction of harvested wood becomes a long-lived product.

Authors of a case study of mixed forest in central Germany (Profft *et al*., 2009) conclude that, under present conditions, there is a slight carbon benefit to leaving the forest standing rather than harvesting for HWP, but that there is scope to swing this balance by increasing the average lifespan of HWP. The authors also note that substitution effects and wood recycling (not considered in their study) would further shift the balance in favour of harvesting and conclude that the carbon mitigation potential of harvesting may exceed that of unharvested forests under favourable site conditions (Profft *et al*., 2009).

If harvest does take place, there is some consensus that long-lived HWP offer a more effective route to climate change mitigation than bioenergy (CCC, 2018b; Geng *et al.,* 2017; Röder *et al.,* 2019). The Committee on Climate Change (CCC, 2018b) recommend that wood should be prioritised for use as a construction material, with the remaining resource used for bioenergy with carbon capture and storage (which, note, is still a mainly theoretical technology at the scale required), but that current uses of bioenergy should be phased out. Röder *et al.* (2019) see a role for bioenergy in providing a market for products that are not suitable for HWP (for example, diseased or damaged trees), thus supporting the economic viability of sustainably-managed forests. DECC (2012) find that the optimal use of forest products in climate change mitigation terms is using small roundwood and sawlogs as a source for HWP (with effective recycling and low-carbon end-of-life disposal an integral part of this), with bark and branchwood produced as co-products used as a source for bioenergy.

6.3 Harvested wood products

Harvested Wood Products (HWP) represent a carbon store in and of themselves, and their use can lead to avoided emissions if it displaces more carbon-intensive alternatives. The current section considers each of these factors in turn.

Carbon stocks in HWP tend to be small compared with the carbon stocks of forests but nevertheless form a significant part of the overall carbon budget of the forest sector (Geng *et al.*, 2017). To increase the overall carbon stored in the system, the lifespan of carbon stored in HWP must exceed that stored in the forest itself if it had remained unharvested (Profft *et al.,* 2009; Law *et al.,* 2014). Numerous studies have attempted to estimate residence time of carbon in different HWP, but there is significant uncertainty in these estimates (Valatin, 2008; Prada *et al.,* 2016) (see Table 3). One analysis found that for the UK, an estimated 30-40% carbon remains in storage in HWP 30 years after land clearance as a fraction of initial above-ground biomass (Earles *et al.,* 2012). The service life of hardwood products tends to be longer than that of softwoods (Valatin, 2008) – however, currently the majority of UK hardwood is used for fuel (81% in 2017) (Forestry Commission, 2018a). Currently, less than 20% of wood harvested in the UK is used as construction timber (Table 3). Any significant shift from short-lived to long-lived HWP would present considerable challenges in terms of current production and consumption patterns. It is also important to note that reducing UK production of short-lived HWP could result in increased imports, shifting the impacts on forest carbon stocks to other countries (Burton *et al.,* 2018; Brainard *et al.,* 2009).

Wood in landfill may be an important long-term carbon store (although also a source of methane); therefore method of final disposal (landfill or burning, with or without energy capture) is important for calculating overall emissions (UNFCCC, 2003; Matthews *et al*., 2014). The carbon benefit of HWP can be significantly enhanced by using more HWP in end uses with long service lives, increasing reuse and recycling of HWP, and using methane produced from decomposing HWP in landfills to generate energy (Geng *et al.*, 2017; Brunet-Navarro, 2018; DECC, 2012; Matthews *et al*., 2014). The Committee on Climate Change (CCC, 2018b) estimate that the use of wood in construction in the UK currently results in over 1 $MtCO_2$ per year being stored in new UK homes, and that increasing timber in construction could treble this figure by 2050.

Tree species affect end use of HWP. Sitka spruce, for example, provides the majority of timber for the wood processing industry (Forestry Commission, 2017). Sitka spruce wood is classified as 'non-durable to slightly durable' and requires treatment with preservatives for many uses, although it is classified as being difficult to treat. This affects its longevity as a harvested wood product. Historically, the main use of roundwood from Sitka spruce was pit props, but with the decline of mining in the UK this market has disappeared and alternative uses of the roundwood are still being developed. High-density Sitka sawlogs are used in structural timber. Sitka wood is used for pallets, but as it has lower bending strength and wood density than other British timber species used for pallet production, it is used for lightweight pallets only. Sitka is used to produce a variety of fencing components and in various types of wood-based panels (Moore, 2011).

The way in which a forest is managed also impacts the expected lifespan of the HWP it yields. For example, one study of mixed forests showed that thinning 'from above' (removing dominant trees to reduce crowding in the main canopy) resulted in HWP with a longer average lifespan than thinning 'from below' (removing less competitive trees) (Profft *et al.,* 2009). In a study of sweet chestnut coppice plantations, introducing thinning resulted in a higher proportion of the wood at final harvest being suitable for long-lived HWP (Prada *et al.,* 2016). Sitka spruce plantations in the UK are commonly thinned to concentrate growth on a smaller number of better-quality trees. This increases the proportion of the standing volume of trees that reaches the minimum size for sawlogs (Moore, 2011). The intensity and timing of thinning is an economic decision – changing market conditions (for example changes in demand for bioenergy) can result in changes to thinning regimes, which could have unintended consequences for carbon sequestration. In Sitka spruce, there is a negative relationship between growth rate and wood density (Moore, 2011), so management to promote fast growth may result in wood that is less suitable for long-lived HWP. Similarly, shorter rotations produce proportionally less mature wood, meaning that timber cut from trees grown in a short rotation have reduced strength and elasticity (Moore, 2011). Longer rotations thus tend to produce a higher share of long-lived HWP (Röder et al., 2019). In a study of the impacts of altering rotation length in European conifer plantations, increasing rotation length up to 70–100 years increased the average carbon stock of wood products, although this increase was an order of magnitude smaller than that of the carbon stock in the forest (Kaipainen *et al.,* 2004).

The use of HWP can reduce the use of carbon-intensive materials such as steel or concrete, which can potentially result in a net reduction in emissions (Geng *et al.*, 2017). Several reports from government and the forestry industry cite displacement effects of HWP (and bioenergy) as an essential part of long-term climate change mitigation strategies (DECC, 2012; Matthews *et al*., 2014; Valatin, 2008). However, estimates of the magnitude of the displacement effect vary widely due to differing system boundaries and assumptions (see for example Valatin 2019). Some authors have found that the substitution effect of HWP has been grossly over-estimated (Law *et al.,* 2014; Harmon 2019). Authors of one literature review contend that under current practices the residence time of carbon in forests is almost always longer than in HWP (Law *et al*., 2014). An examination of the underlying assumptions in existing models concluded that the long-term benefits from the substitution effect may have been overestimated 2- to 100-fold (Harmon, 2019). Three key assumptions implicit in many existing carbon calculations are examined:

1. The carbon displacement value of HWP remains constant over time. This is not realistic given continued technological developments altering the efficiency of processes in the energy and construction sectors.
2. The displacement is permanent, i.e. any reduction in fossil fuel use achieved by using HWP means this carbon will stay in the ground forever. Given that fossil fuels are a finite resource, it seems far more likely that this carbon will simply be used by other sectors, an effect known as carbon leakage.
3. There is no relationship between building longevity and substitution longevity. This is based on the assumption that, when a wooden building reaches the end of its lifespan, by default it will be replaced with a non-wooden building. Using this scenario as a baseline means that all ongoing use of HWP can be counted as a carbon

saving, and the carbon store in HWP never saturates. The author's calculations show that the shorter the average lifespan of buildings the smaller the total amount of carbon displaced over the long term. Many models effectively assume that buildings have an infinite lifespan, which is clearly unrealistic and overestimates the amount of carbon displacement.

Applying a more realistic set of assumptions led to the conclusion that maximizing harvest yields using short rotation forestry may not lead to the greatest overall climate mitigation. More focus should be given to maintaining the amount of carbon displaced, reducing carbon leakage and increasing the longevity of buildings (Harmon, 2019).

6.4 Forest-based bioenergy

Using wood for energy means that carbon that would otherwise have been stored in living biomass or HWP, or slowly released from decaying biomass, is released into the atmosphere much more rapidly (Egnell *et al.,* 2016; Whittaker *et al.,* 2011). The scope for climate change mitigation from bioenergy arises from its potential to displace fossil fuel use. Carbon emitted from burning biomass is part of the forest-atmosphere carbon cycle, so as long as forests regrow the total amount of carbon in the atmosphere remains approximately constant over the long-term (apart from some use of fossil fuels in harvesting and processing biomass) (Ter-Mikaelian *et al.,* 2014). This is by contrast with burning fossil fuels, which permanently increases the amount of carbon in the global carbon cycle (Geng *et al.,* 2017; Ter-Mikaelian *et al.,* 2014). However, a key question that is often overlooked is how long it takes to reach net atmospheric carbon reduction. This is significant because for as long as atmospheric carbon levels are elevated the effects will be felt in terms of climate change and direct biological impacts (Ter-Mikaelian *et al*., 2014). The possible risk of reaching climate 'tipping points' is also increased (Brack, 2017). From a policy perspective, long payback periods are not compatible with reaching short term climate targets (Norton *et al*., 2019; Brack, 2017).

Burning wood results in an immediate increase in atmospheric carbon dioxide: a carbon 'debt' that is repaid over a certain period of time as forests regrow (Norton *et al.,* 2019; Yan, 2018; Geng *et al*., 2017). The length of this payback period depends largely on the initial feedstock (Norton *et al.,* 2019). Where forestry or mill residues are used, the carbon debt may be repaid in a matter of years, whereas the felling of living trees results in a debt that may take decades to centuries to repay (Norton *et al.,* 2019; Ter-Mikaelian *et al.,* 2014; Brack, 2017). In some cases the initial loss of carbon will never be recovered (Norton *et al.,* 2019). By comparison, renewable technologies such as solar and wind power produce net carbon dioxide savings within months to a few years (Norton *et al.,* 2019).

In the UK and Ireland much of the primary residue material from harvest is currently left on the forest floor as it is more difficult and expensive to process than other fuel sources. Stem tips and branches are sometimes purposely left as 'brash mats' that help to protect the forest floor from damage by heavy machinery (Whittaker *et al.,* 2011; Murphy *et al.* 2014). If demand for biomass outstrips the supply of secondary and tertiary residues, the initial response of foresters tends to be an increase in harvest intensity, extracting a higher proportion of the biomass in the forest. Branches, tops, stumps, and ultimately whole trees can be used to supply biomass for energy. As well as reducing the size of the forest carbon store, this can reduce the ongoing rate of carbon draw-down in the forest (Egnell *et al.,* 2016; CCC, 2018b). This may or may not be counterbalanced by the increased displacement of fossil use made possible by the increased biomass yield (Whittaker *et al.,* 2011; CCC, 2018b). In a counterfactual situation where this biomass is not harvested but left to decay in situ, estimates of its residence time vary substantially (Brack, 2017), ranging from 2 years to 500 years depending on specifics of the biomass and site conditions, although most field studies indicate that woody debris does not add significant amounts of carbon to the soil in the long run (Egnell *et al.,* 2016). It has been argued that coarse woody biomass like stumps and stemwood from long-rotation forestry should not be used for bioenergy because the payback time is long (as it takes a long time for these types of wood to regrow, they take a long time to decompose once dead, thus forming a long-term carbon store, and harvesting them involves considerable soil disturbance) (Egnell *et al.,* 2016; CCC, 2018b; Brack, 2017).

Intensive management of plantations harvested for bioenergy may reduce the length of the payback period (Ter-Mikaelian *et al*., 2014) (potentially at the cost of significant impacts on biodiversity, discussed elsewhere in the current report). The Committee on Climate Change (CCC, 2018b) points out that fast-growing forests provide biomass more rapidly than slower growing forests, enabling more cumulative emissions to be displaced from the energy system over a given time period. However, reducing harvest rotation periods (in the absence of other changes such as restocking with faster-growing trees) reduces the average level of carbon stocks in a forest over the harvest cycle (CCC, 2018b; Prada *et al*., 2016).

It is sometimes argued that if the forest or landscape from which the biomass is sourced has stable or increasing carbon stores overall, there is no carbon debt and bioenergy is effectively carbon neutral, apart from the consumption of fossil fuels during harvesting and handling the biomass (Whittaker *et al.,* 2011; studies referenced in Geng *et al.*, 2017; Matthews *et al*., 2014). This argument is flawed because it fails to take into account the scenario where no harvesting takes place (Ter-Mikaelian *et al.,* 2014). The equilibrium level of carbon in a forest subject to harvesting is lower than that of an unharvested forest (Geng *et al.*, 2017). Where (as is common practice) stands are harvested before they reach the stage of maximum above-ground biomass (i.e. while they are still actively growing), there is a lost opportunity for future carbon draw-down, further increasing the carbon debt (Ter-Mikaelian *et al.,* 2014).

Estimates of overall emissions reductions (from both bioenergy and HWP) depend to a great extent on the chosen counterfactual scenario, which could be for example leaving the forest to grow or harvesting it for HWP only, while meeting energy demands with fossil fuels (coal, oil, gas) or alternative energy sources (such as renewable energy) (Geng *et al.*, 2017; CCC, 2018b; Ter-Mikaelian *et al.,* 2014). Currently, replacing coal with biomass for electricity generation is likely to significantly increase emissions per unit of electricity generated. This is a result of numerous factors including the lower energy density of wood compared to coal, emissions along the supply chain, and less efficient conversion of combustion heat to electricity (Norton *et al.*, 2019; Brack, 2017). Replacing gas with biomass is even less favourable (Brack, 2017). When considering longer timeframes, the counterfactual scenario must take account of possible developments such as improved efficiency or decarbonization in the energy and construction sectors and the commercialisation of carbon capture and storage technology (Valatin, 2008). The Committee on Climate Change (CCC, 2018b) found that, depending on how it is produced, forest bioenergy can correspond to a range of greenhouse gas outcomes, higher or lower than fossil fuel equivalents.

Case study 1: the importance of timeframe

In a model of a temperate deciduous forest harvested for bioenergy (Yan, 2018), the length of the payback time was found to increase with the length of rotation (in this example, the payback time was 18, 25 and 35 years when the rotation was 30, 50 and 100 years respectively). When considering net carbon emissions over a short time horizon of 20 years, harvesting had a negative effect (i.e. increased net emissions), because the carbon debt had not yet been paid back. This negative impact was greatest when the forest was harvested on a rotation of 50 years because the fastest rate of growth occurs when the stand is 50 – 70 years old (i.e. cutting down the forest just before it enters its period of highest growth rate sacrifices the most sequestration benefit). However, looking at net emissions over a longer period resulted in a different picture. Once the carbon debt of harvest has been paid off there is annual net sequestration. This takes place at a slower rate in older forest stands, but nevertheless delaying harvest had a net positive effect (reduced net emissions) when the time horizon considered was 100+ years. Further modelling by the same author of both deciduous and coniferous plantations found that reducing harvest intensity ensures fast re-growth of biomass. The author's overall conclusions were that harvest has a negative impact on sequestration by forests, but increasing tree growth rate, lengthening rotation period and reducing harvest intensity could reduce this impact.

Case study 2: defining the desired outcome when comparing forestry systems

A study in 2019 evaluated the greenhouse gas mitigation potential of different supply chains producing wood pellets for large-scale electricity generation in the UK (Röder *et al.,* 2019). The supply chains considered were intensively-managed loblolly pine plantations in the USA on a rotation of 25 years, extensively-managed boreal forests in Canada clearfelled at 70 years and allowed to naturally regenerate, and eucalyptus plantations in Spain coppiced at 16 years and clearfelled and replanted at 32 years. The American and Canadian forests produced HWP with bioenergy as a secondary product, while the Spanish eucalyptus was managed purely for bioenergy. Lifecycle assessments were carried out encompassing all aspects of forest carbon fluxes and the processing, use and disposal of wood for energy and HWP. Greenhouse gas balance was assessed over a period of 100 years. A detailed discussion of the results is beyond the scope of the current paper, but this case study serves to illustrate the complexity of analysing climate change implications of forest management. The Canadian boreal forest had the lowest emissions associated with its management and therefore produced the lowest emissions per unit of energy yielded. The USA and Spanish plantations sequestered much more carbon over the same timeframe compared to the slow-growing Canadian boreal forest. In the USA and Canadian forest systems, the total system's carbon stock showed a net increase from rotation to rotation because the carbon pool in HWP constantly accumulated in HWP with a longer lifespan. By contrast, because all of the wood harvested from the Spanish eucalyptus was burned, total carbon stock between successive rotations did not change. Both the USA and Spanish systems soon reached a point where the cumulative emissions from bioenergy use exceeded the cumulative carbon sequestration of the plantations (2-4 years for the American pine plantation and 8 years for the Spanish eucalyptus). This also happened for the Canadian boreal forest, but in this case it took 72 years. The authors conclude that bioenergy only contributes to net carbon savings if the carbon sequestration rate can be maintained at a high level throughout the forest and electricity production lifetime. This may be achieved by expanding the forest area, but this is clearly a finite option.

6.5 Summary and conclusions

As stated in the introduction to the current section, the focus is on scientific understanding of carbon implications of the fate of harvested wood in the UK. This section does not look at biodiversity implications, international forestry, the impacts of possibly future technologies or the complexities of policy design and implementation (governing for example forest management practices, carbon accounting and offsetting). The following is a summary of the key findings.

- The fate of harvested wood is an important driver of the greenhouse gas balance of the overall forestry system. Harvesting reduces the equilibrium level of carbon in the forest but can provide long-term carbon storage opportunities outside the forest, as well as potentially reducing the use of fossil fuels and non-wood products.
- To gain an accurate picture of the carbon implications of harvesting forests, HWP and bioenergy life-cycle carbon analysis needs to be integrated with forest carbon balance analysis. Many previous studies have focused only on one or the other of these.
- Some authors have concluded that the no-harvest scenario is preferable to harvesting for bioenergy or HWP, i.e. that greater climate change mitigation can be achieved by leaving the forest standing and thus increasing its carbon stores. If harvest does take place, there is some consensus that long-lived HWP offer a more effective route to climate change mitigation than bioenergy
- Harvested Wood Products (HWP) represent a carbon store (although this is small compared to the carbon stock of the forest). Numerous studies have attempted to estimate residence time of carbon in different HWP, but there is significant uncertainty in these estimates. Both tree species choice and forest management affect the average lifespan of HWP. The carbon benefit of HWP can be enhanced by using more HWP in end uses with long service lives, increasing reuse and recycling of HWP, and using methane produced from decomposing HWP in landfills to generate energy.
- The use of HWP can reduce the use of carbon-intensive materials such as steel or concrete, leading to climate benefits from carbon displacement. However, estimates of the magnitude of the displacement effect vary widely and some authors have found that commonly cited figures are gross over-estimates.
- Burning wood for energy releases carbon to the atmosphere. Unlike burning fossil fuels, this does not increase the total amount of atmospheric carbon in the long term. However, forest-based bioenergy cannot be considered carbon neutral because the payback time until the carbon is reabsorbed can be very long, particularly when living trees are felled for biomass. Harvest residues have a shorter payback time but increasing their use can have implications for the forest’s continued ability to grow and absorb carbon. It is often argued that where carbon stocks are constant over a landscape scale (i.e. some forest stands are felled while others continue to grow) there is no carbon debt. However, this ignores the scenario where no harvesting is carried out, when the carbon equilibrium of the landscape would be higher.
- Replacing coal or gas with biomass for electricity generation is likely to significantly increase emissions per unit of electricity generated.

- By comparison, renewable technologies such as solar and wind power produce net carbon dioxide savings within months to a few years.

Table 3: overview of uses of UK-grown wood and estimated lifespan of HWP

Uses of harvested UK wood in 2018 (green tonnes)	**Estimates of lifespan of HWP**
Sawmills *Overall proportion of UK wood: 56%* Softwood 6 426 000 Hardwood 67 000 Of which*: 33% used in construction 36% used for fencing 24% used for packaging & pallets 7% used for other markets	Half-life of 30 – 50 years[1] Service life of 35 years[2] Half-life of 70 – 100 years (construction timber)[3] Service life of 15 years (fencing)[4] Half-life of 6 years (packaging and pallets)[3] Variable
Wood-based panels *Overall proportion of UK wood: 11%* Softwood 1 210 000 Hardwood 1000	Service life of 25 years for particleboard[2]
Fencing *Overall proportion of UK wood: 2%* Softwood 273 000 Hardwood 0	Service life of 15 years[4]
Pulp mills *Overall proportion of UK wood: 4%* Softwood 486 000 Hardwood 67 000	1 -2 years[1] 2 years[2]
Woodfuel *Overall proportion of UK wood: 23%* Softwood 1 900 000 Hardwood 700 000	Zero
Other uses *Overall proportion of UK wood: 2%* Softwood 174 000 Hardwood 66 000	Variable
Exported *Overall proportion of UK wood: 2%* Softwood 264 000 Hardwood 0	Variable

Sources: 1. IPCC, 2003; 2. Brown *et al.,* 2018; 3. UNFCCC, 2003; 4. Moore, 2011. Estimated proportions of uses of wood are adapted from Forestry Statistics 2018, tables 2.5 and 2.6 (Forestry Commission, 2019).

*these percentages apply to the 86% of timber processed by larger sawmills in 2018, defined as those that process over 25000 m^3 of sawn wood annually

Appendix A– literature review methods

My criteria for including research in the report were as follows:

- Relevance to UK: research carried out in temperate forests, mainly Europe but some USA if using tree species relevant to UK forestry.
- Published since 2009 (or seminal work or research not updated since original paper)
- Peer reviewed and/or from one of the organisations listed below.

The majority of the papers referenced in this report are freely available online – due to resource constraints it was not possible to follow up all of the harder-to-obtain references.

To find research for inclusion I carried out searches on Science Direct using the following terms: (Wood OR Forest OR Plantation OR Afforestation) AND (Carbon OR "Greenhouse gas" OR Climate OR "Global warming" OR Emission OR Sequestration OR Sink OR Mitigation). I limited the results to papers published from 2009 onwards. This generated nearly 200,000 results. I sorted the results by decreasing relevance and worked through the list, reading titles and abstracts to generate a shortlist of papers for inclusion in the report. I continued until three consecutive pages of results returned zero shortlisted papers. I then repeated a similar exercise using Google Scholar and Google.

I also searched the websites of the following organisations for grey literature: Defra, IUCN, Forest Research, Committee on Climate Change, Scot Gov, DECC, BEIS, IPCC, IUCN, RSPB, ClimateXChange.

I also found some relevant research in the bibliographies of the papers I was reviewing. Specialists from the RSPB suggested additional research, particularly covering biodiversity issues.

The section on fate of harvested wood is not a systematic literature review of this subject area. It is based on papers found during the literature review carried out for the main report plus a small number of key papers suggested by the RSPB.

Appendix B – forestry standards in the UK

The information provided in this appendix is intended as background for readers who may not be familiar with these standards. It is not the aim of the current report to critically assess current forestry policy or its implementation. The UK Forestry Standard (UKFS) sets out the approach of the UK governments to sustainable forest management, and applies to all UK forests (Forestry Commission, 2017). It refers to both legal requirements and 'good practice' requirements, which in theory forest owners must meet in order to receive certification or government grants. Requirements cover a range of topics including management planning; maintaining forest productivity (defined to include both timber production and other ecosystem services); creating and maintaining a varied forest structure; dealing with pests and diseases; use of chemicals; conserving and enhancing biodiversity; adapting to and helping mitigate climate change; preserving the historic environment and landscape character; providing social benefits; and protecting soil health and water quality. Some of the specific requirements and guidelines most relevant to the current report are:

- Ensure the removal of forest products does not deplete soil carbon over the long term;
- Forest and woodlands should be designed to achieve diverse habitat, species and ages of trees appropriate to scale context and ecological potential of site.
- For the progressive restructure of even aged forests.
- For enhancement of visual, cultural and ecological value of the landscape.
- Forests and woodlands should be managed to conserve or enhance biodiversity.
- Consider implications of woodland creation and management for biodiversity in the wider environment, including for forest habitats, open habitats and ecological connectivity.
- Management should contribute to long-term climate change mitigation through the net capture and storage of carbon in the forest ecosystem and in wood products.
- Consider alternatives to clearfell systems, such as continuous cover forestry, where suitable sites and species combinations allow and management objectives are compatible;
- Where woodlands are managed for timber production, maximise carbon sequestration through efficient management, consistent with the output of durable products;
- Consider the potential for woodfuel and energy crops within the sustainable limits of the site;
- Avoid removing stumps unless for tree health reasons or where a risk-based assessment has shown that adverse impacts can be reduced to acceptable levels;
- Minimise soil disturbance.
- Protection or enhancement of forest soil physical, chemical and biological properties.
- Avoid damage to soil structure and function through forestry operations.

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