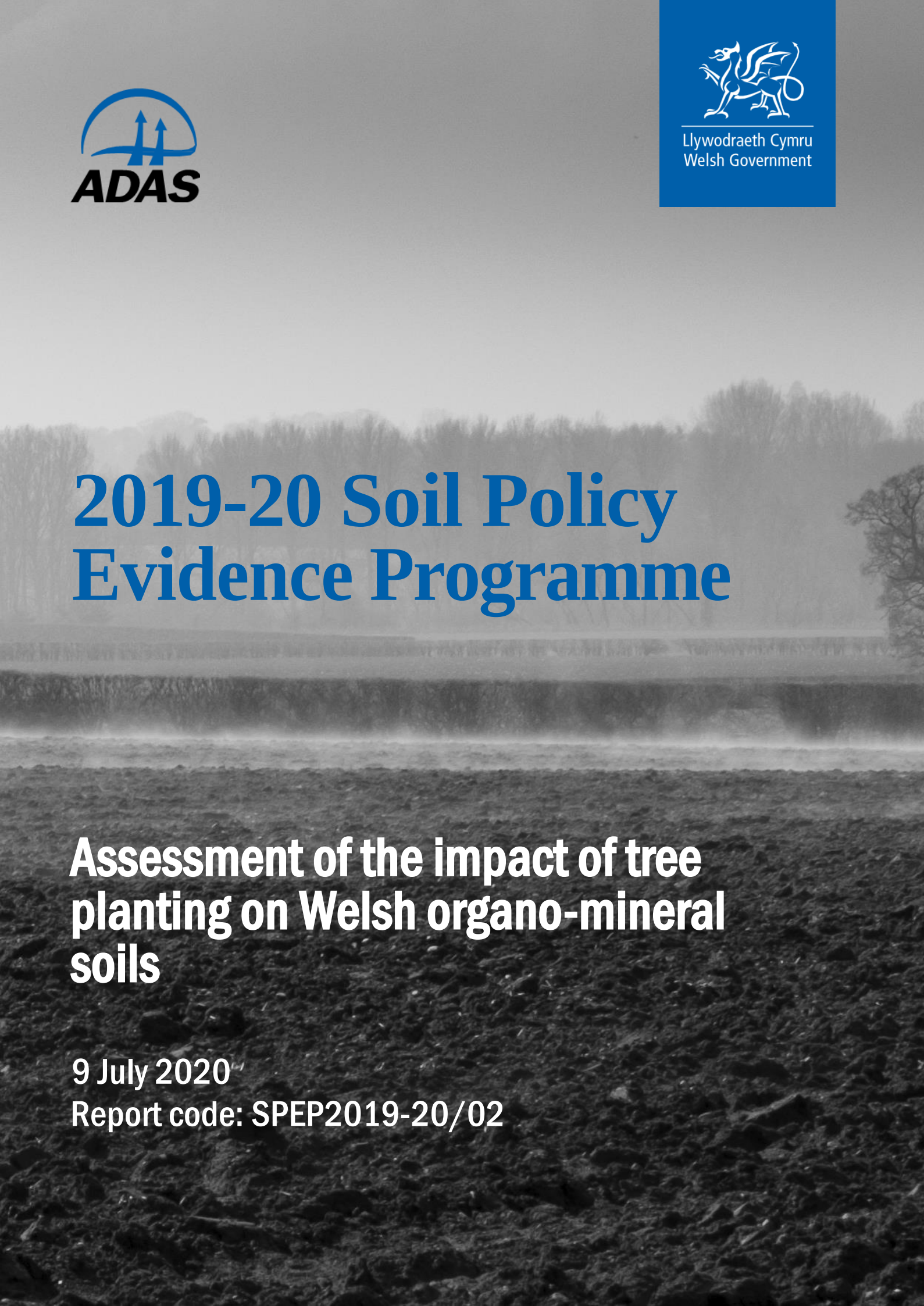

ADAS

Llywodraeth Cymru
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2019-20 Soil Policy Evidence Programme

Assessment of the impact of tree planting on Welsh organo-mineral soils

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Introduction

Wales has committed to the creation of substantial areas of new woodland to meet net zero carbon targets for 2050 (Welsh Government, 2018). To meet the legal requirement of the 'Wales 80%' target by 2050 (reduction in carbon emissions to 80% of pre-1990 levels), it has been suggested that 2000 ha of woodland should be planted each year from 2020 to 2030 followed by a further increase in annual planting from 2030 to 2050. Identifying the most suitable land for new tree planting is therefore an important challenge for decision makers particularly given the conflicting pressure upon land use for a multitude of different purposes including food production, biodiversity provision and commercial development.

Soils are an integral part of the terrestrial ecosystem and provide a number of important functions including carbon storage, climate regulation, water regulation, nutrient cycling and support for biodiversity (Gregory *et al.*, 2015). It is estimated that of the total UK biocarbon stock (4,266 MtC), 94.2% (4,019 MtC) is stored within soil whilst 5.8% (247 MtC) is stored in vegetation (ONS, 2016). Land use and land management practices strongly affect soil functions including carbon storage and have the potential to impair soil functioning and increase susceptibility to erosion and degradation. Therefore, to select suitable sites for forest creation it is critical that the influence of land use change upon soil functions is fully assessed.

Organo-mineral soils are defined as those with a peat topsoil (i.e. one that contains more than 20% organic matter) which is less than 40 cm deep. These soil types cover *c*. 20% of the total land area and contain *c*. 25% of the carbon stored in Welsh topsoils (Bol *et al*., 2011). Much of Welsh organo-mineral soils are in upland areas (> 600 m asl.) where low temperatures and high rainfall restrict agricultural production More than 97% of Welsh organo-mineral soils are classed as LFA land (less favourable for agriculture).

Environmental benefits of tree planting are site specific and it is well established that afforestation in unsuitable suites can negatively impact aspects of environmental quality including soils, water and biodiversity (NCC, 2020). Both the UK Forestry Standard and UK Woodland Assurance Standard state a requirement that there should be no new planting on deep peat soils (>50 cm depth) or where planting would compromise adjacent peat bog habitat (Forestry Commission, 2017; UKWAS, 2018). This is because forestry activities such as cultivation, trafficking, draining and drying increase carbon oxidation from peatland soils leading to significant carbon losses and degradation of peatland habitats (Campbell & Robson, 2019). Organo-mineral soils are an important carbon stock and whilst they store less carbon per unit area than deep peats, they generally hold more carbon than mineral soils (Bol *et al.*, 2011). The ECOSSE project estimated that In Wales deep peats and organo-mineral soils together cover around 20% of the land area and that these soils respectively account for 31% and 19% of the total soil carbon stock to the depth of the profile (Scottish Executive, 2007). In contrast mineral soils hold 50% of the total soil carbon despite occupying 80% of the land area (Scottish Executive, 2007). It is therefore important to fully consider how woodland creation and forestry activities may impact the functions provided by organo-mineral soils particularly carbon storage and the susceptibility of these soils to degradation.

Objectives

The aim of this report was to review the current available evidence to assess the impact of planting trees upon Welsh organo-mineral soils for carbon storage and soil functions.

Government policies for woodland creation

Globally, the creation of forest is considered to be an important tool for mitigating anthropogenic climate change and has thus received much recent attention (Bastin *et al.*, 2019; CCC, 2020). Forest creation can potentially play an important role in mitigation of greenhouse gas emissions for the UK (Read *et al*., 2009) by removing carbon dioxide from the atmosphere and storing it in above and below ground biomass. The creation of new woodland and associated benefit for carbon sequestration could

provide an important contribution to the UK's commitment to achieve net zero carbon by 2050 (CCC, 2020).

Woodland currently occupies approximately 13% of the UK land area therefore achieving the UK target of 19% tree coverage by 2050 requires a significant increase in afforestation. A 6% increase in UK woodland area equates to the creation of around 1.5 million hectares of additional woodland and requires the annual planting of 50,000 ha from 2020 to 2050. Woodland area targets do not include canopy cover of trees outside areas defined as woodland, for example hedgerows, field trees, city trees, riverside trees or wood pasture.

The Committee on Climate Change (CCC) recently assessed the contribution that changes in land use and agriculture could make to achieving net zero emissions by 2050 in the UK (Figure 1) (CCC, 2020). Their report suggested that increasing woodland to 17% of the UK land area in addition to improved woodland management could enable woodlands to sequester 14 $MtCO_2e$ annually by 2050. Furthermore there could be potential to sequester an additional 14 $MtCO_2e$ annually through the use of wood products harvested from forestry in other areas of the economy (CCC, 2020). Overall this report shows that changes to forestry and woodland management within the UK could make an important contribution to the emission savings needed to achieve net zero. In total the suggested changes to forestry related activities (creation and management of forests, management of forestry related products and agroforestry) were estimated to be able to contribute to savings of 34 $MtCO_2e$. This is almost half of the total savings of 69 $MtCO_2e$ required to reduce GHG emissions in line with 2050 net zero targets (CCC, 2020).

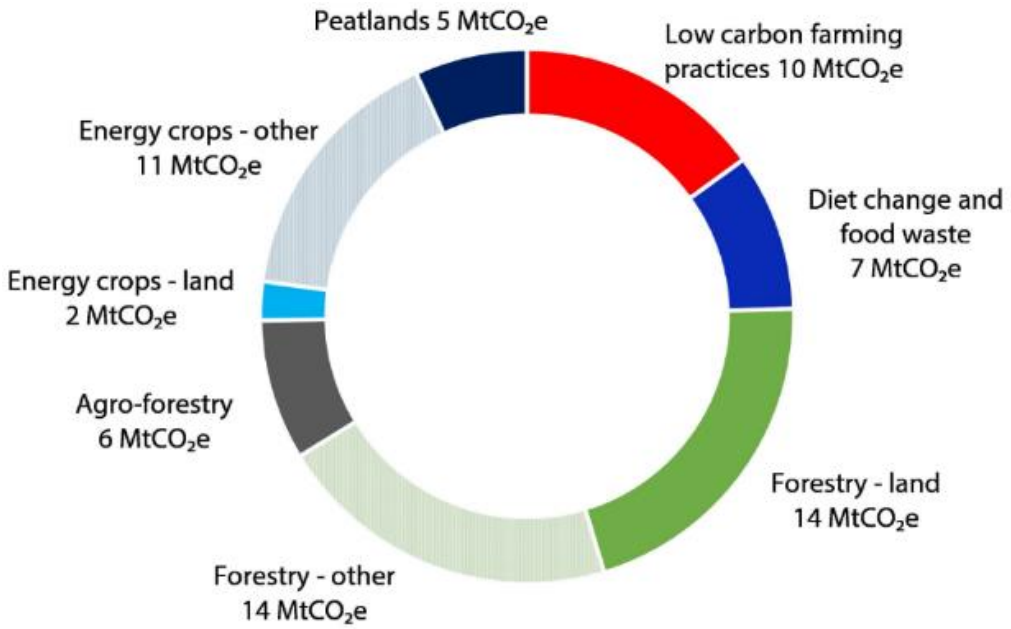


Figure 1. Contribution of greenhouse gas (GHG) emissions by agriculture and land use changes to the total GHG emission reduction needed to achieve net zero by 2050. Savings are in comparison to business as usual GHG emissions. 'Energy crops - other' and 'Forestry - other' refer to GHG savings from the use of harvested products. From (CCC, 2020).

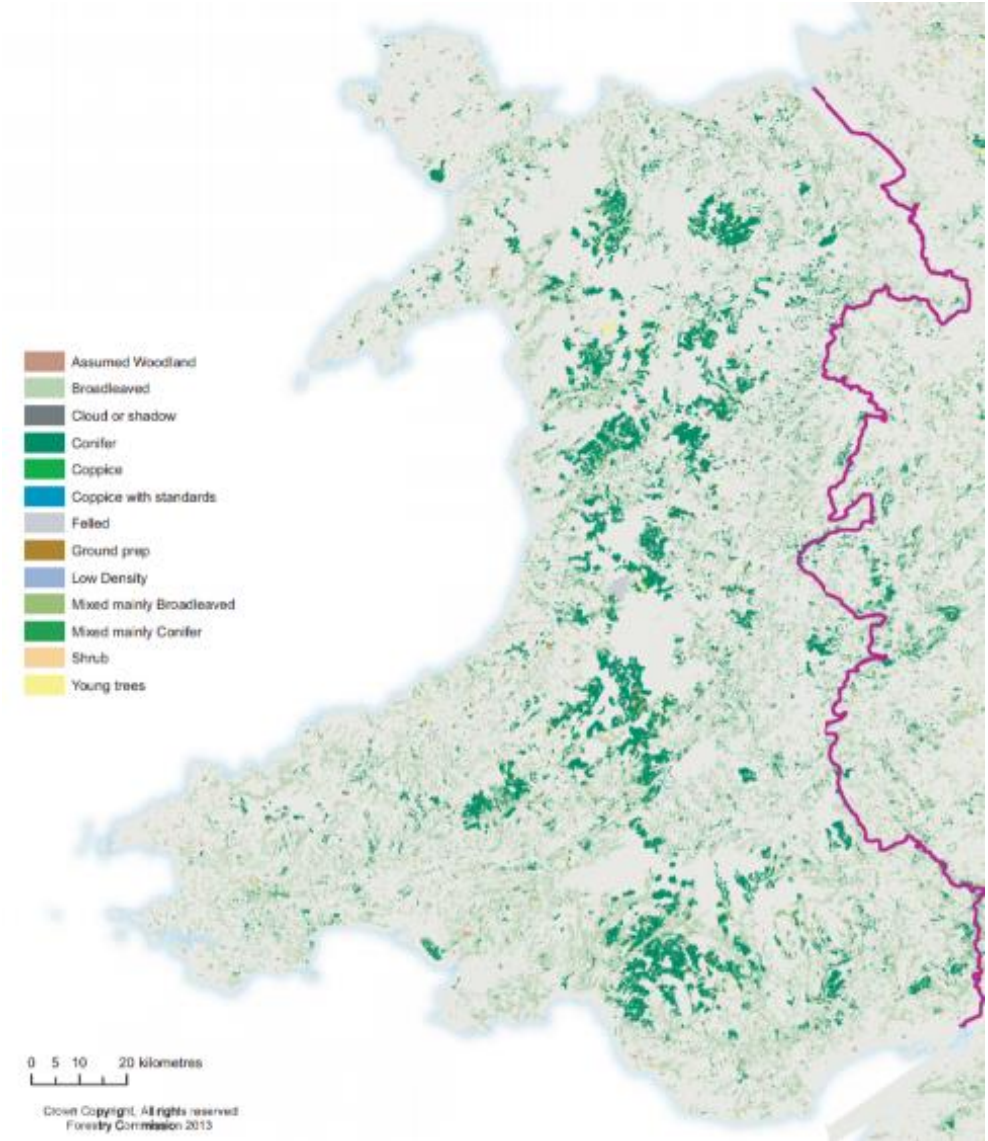
Currently forestry covers 15% (309,000 ha) of the land area of Wales of which 8%[1] is native woodland (Forest Research, 2019a). The Welsh Government have committed to reducing carbon emissions as part of the Environment (Wales) Act 2016. As part of a strategy to reduce carbon emissions to 80% of the pre-1990 emission levels, the Committee on Climate Change have proposed a target of planting

[1] 1.2% of the land area - 3,700 ha

2000 ha of additional woodland each year from 2020 to 2030 after which the area of planting must be further increased (Welsh Government, 2018). In comparison, to achieve the target of net zero carbon by 2050 would require at least 4000 ha of new woodland to be planted each year from 2020 to 2030 (Welsh Government, 2018). Both of these scenarios present a substantial increase from the current rate of new planting in Wales which between 2010 and 2019 averaged at 452 ha per year (Forest Research, 2019a). The Committee on Climate Change emphasises that significant policy change will be required to facilitate this increase in afforestation.

Welsh woodlands

At present 15% of land in Wales is classed as woodland using the definition of woodland as "stands of trees with a minimum area of 0.5 ha and a canopy cover of at least 20%" (Map 1) (Forestry Commission, 2019). Of the 309,000 hectares of Welsh woodland, 51% (158,000 ha) are broadleaf and 49% (172,000 ha) coniferous (Forest Research, 2019a). The most widely planted broadleaf and coniferous tree genus are detailed in Table 1. In 2018 96,000 ha of woodland in Wales were reported as farm woodland (Forest Research, 2019a). The total forest soil carbon stock (to 1 m depth) for Wales is estimated to be 192 $MtCO_2$ (52 MtC). This amount constitutes 8% of the total forest soil carbon stock (to 1 m depth) within Great Britain which is estimated to be 2302 $MtCO_2$ (627 MtC) for 2.66 Mha of forest (Vanguelova *et al.*, 2013).



Map 1. Distribution of Welsh woodland by forest type (National Forest Inventory, 2011).

Table 1. Stocked woodland area in Wales for conifers and broadleaves by area of principal tree types. Species names indicate specific species which are particularly widely planted rather than multiple species within a genus. Data from Forestry Commission (2014a,b).

Principal Conifers:	Thousand hectares	Principal Broadleaves:	Thousand hectares
Sitka spruce (*Picea sitchensis*)	77	Oak (*Quercus*)	26
Larches (*Larix sp.*)	20	Ash (*Fraxinus*)	19
Douglas fir (*Pseudotsuga menziesii*)	9	Hazel (*Corylus*)	14
Norway spruce (*Picea abies*)	8	Birch (*Betula)*	12
Lodgepole pine (*Pinus contorta*)	4	Willow (*Salix*)	11
Scots pine (*Pinus sylvestris*)	3	Alder (*Alnus*)	10
Corsican pine (*Pinus nigra*)	2	Sycamore (*Acer*)	9
Other conifers	5	Hawthorn (*Crataegus*)	8
		Beech (*Fagus*)	6
		Other broadleaves	21
All conifers	129	All broadleaves	137

Welsh organo-mineral soils

Organo-mineral soils

Organo-mineral soils have been broadly described as soils with a surface horizon rich in organic matter which is less than 40 cm thick (in England and Wales), overlying rock or mineral horizons (Scottish Executive, 2007; Bol *et al.*, 2011). It is important to note that soils with an organic layer <40 cm deep are classed as organo-mineral in England and Wales whilst in Scotland soils with an organic layer <50 cm deep may be classed as organo-mineral. Soils are classed as peat where these respective depths are exceeded per country. Organo-minerals soils are also referred to as shallow peats particularly within the forestry literature.

Based on the soil classification of England and Wales (Avery, 1980), Bol *et al*., (2011) identified three categories of organo-mineral soil subgroups. These organo-mineral subgroups are detailed in Table 2 whilst the limiting percentages of organic matter and organic carbon for mineral, organo-mineral and organic soils used by Bol *et al*., (2011) are shown in Figure 2. To be classed as organo-mineral Bol *et al*., (2011) considered that the surface horizon had to be rich enough in organic matter to belong to at least one of the following categories:

(i) >15 cm thick humose topsoil.

(ii) >15 cm thick peaty loam or peaty sand topsoil (<20% organic carbon).

(iii) Peat (loamy, sandy, fibrous, semi-fibrous or amorphous) <40 cm thick starting at or near the surface or <30 cm thick where peat lies directly on bedrock.

A detailed description of the characteristics and distribution of the soil groups presented in Table 2 is provided for England and Wales by the National Soil Research Institute (Cranfield University, 2020).

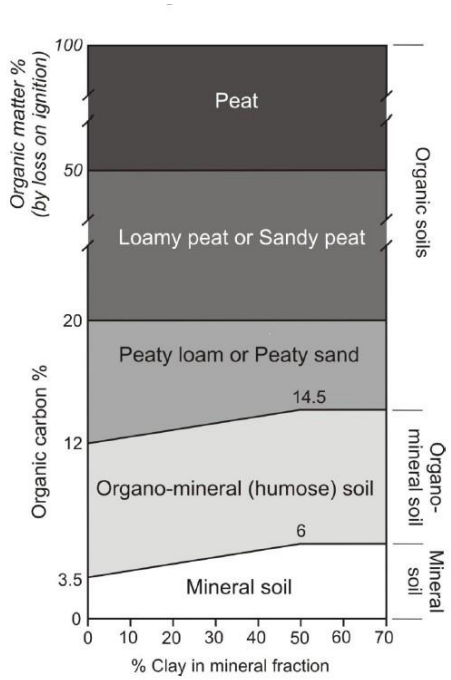


Figure 2. Organic carbon and organic matter content of organic, organo-mineral and mineral soils. From (Bol *et al.*, 2011) based on (Hodgson, 1997).

Table 2. Soil subgroups from the National Soil Map for England and Wales (Cranfield University, 2020) considered to be organo-mineral soils by Bol *et al.*, (2011). Wetness classes refer to categories by Hodgson (1997) (Appendix 1A). Adapted from Bol *et al.*, (2011).

Category	Description	Wetness class	Soil subgroups	Soil subgroup number	Soil associations[2]
1	Freely and moderately well drained with humose or thin (<40 cm thick) peaty surface horizon (topsoil).	I, II	Humic Rankers Humic Rendzinas Humic Brown Podzolic soils Humo-ferric podzols	311 341 612 631	311a Revidge, 311b Skiddaw; 311c Wetton 1; 311d Wetton 2; 311e Bangor 341 Icknield 612a Parc; 612b Moor Gate 631a Anglezarke; 631b Delamere; 631c Shirrell Heath 1; 631d Shirrell Heath 2; 631e Goldstone; 631f Crannymoor
2	Poorly drained Podzols with humose or thin (<40 cm thick) peaty surface horizon	IV, V, VI	Ferric Podzols Typical-gley Podzols Stagnogley Podzols Ironpan Stagnopodzols Humus-ironpan Stagnopodzols Ferric Stagnopodzols	633 641 643 651 652 654	633 Larkbarrow 641a Sollom 1; 641b Sollom; 641c Holme Moor 643a Hollidays Hill; 643b Poundgate; 643c Bolderwood 651a Bellmont; 651b Hexworthy; 651c Earle 652 Maw 654a Hafren; 654b Lydcott; 654c Gelligaer
3	Poorly drained Stagnohumic Gleys, Pelo-alluvial Gleys and Humic Gley soils with humose or thin (<40 cm thick) peaty surface horizon	V, VI	Stagnohumic Gley soils Pelo-alluvial Gley soils Typical Humic-alluvial Gley soils Typical Humic-sandy Gley soils Typical Humic Gley soils Argillic Humic Gley soils	721 813 851 861 871 873	721a Princetown, 721b Onecote; 721c Wilcocks 1; 721d Wilcocks 2, 721e Wenallt 813a Midelney; 813f Wallasea 1 851a Downholland 1; 851b Downholland 2; 851c Downholland 3 861a Isleham 1; 861b Isleham 2 871a Laployd; 871b Hense; 871c Hanworth 873 Ireton

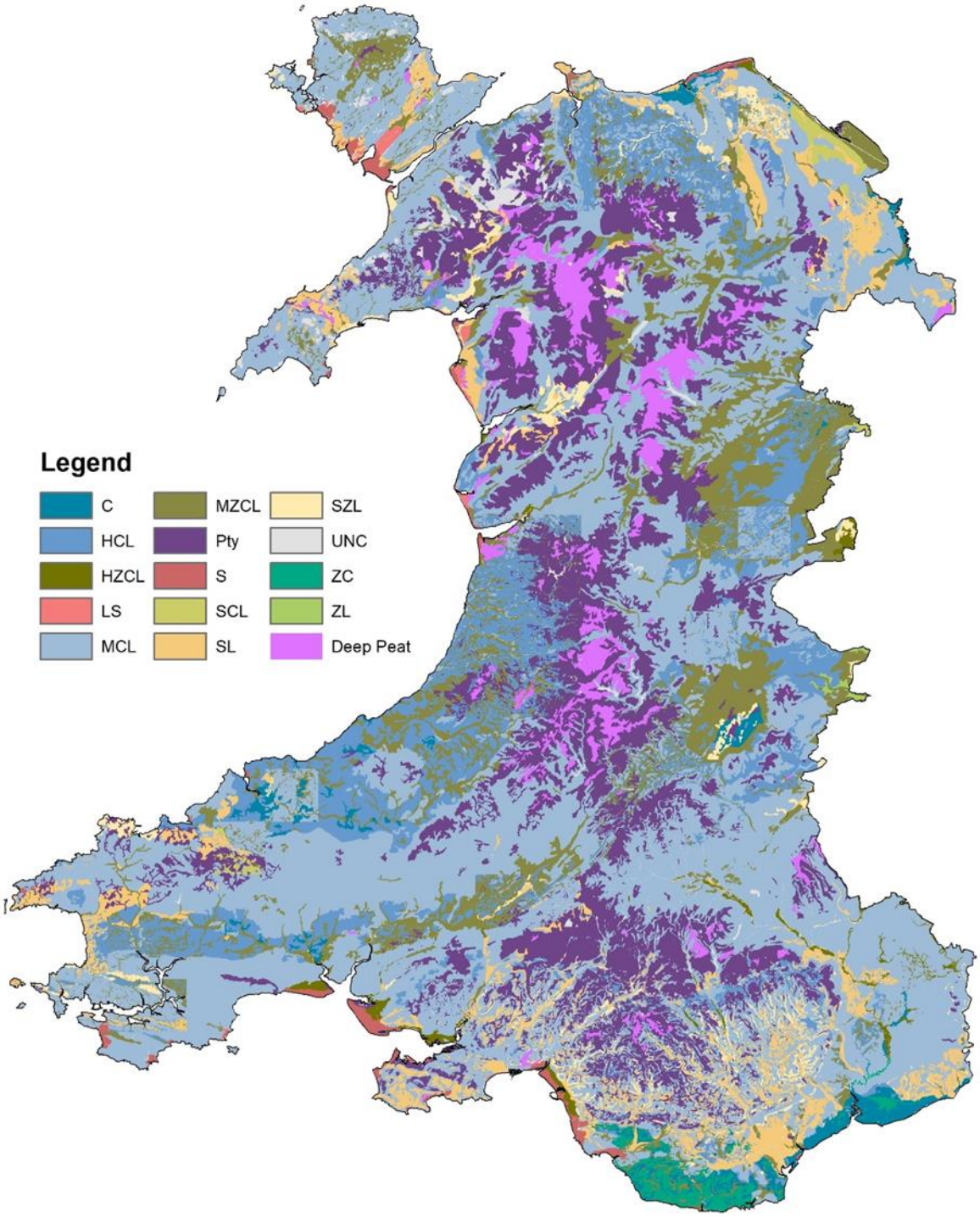
[2] National Soil Research Institute soil associations

Distribution

Organo-mineral soils have been estimated to account for 20.5 % of the land area of Wales based upon National Soil Resources Institute (NSRI) data for land cover of individual soil associations shown in Table 2 (Bol *et al.*, 2011). The majority of Welsh organo-mineral soils, 363,118 ha (*c*. 85 % of total organo-mineral soil area) are located in upland areas including the Cambrian Mountains, Brecon Beacons and Snowdonia, with only 62,900 ha (*c.* 15 % total organo-mineral soil area) situated in lowland areas (Table 3, Map 2) (Evans *et al.*, 2015).

Table 3. Land cover of organo-mineral soils within Wales as reported by Bol *et al.*, (2011)

Organo-mineral soil category	Lowland Wales: Area (ha), % total organo-mineral soil area	Upland Wales: Area (ha), % total organo-mineral soil area	Total land area within Wales (ha)	% of total area of Wales
Well drained	16647 (3.9%)	55998 (13.1%)	72645	3.5
Podzols	4532 (1.1%)	181664 (42.6%)	186196	9.0
Gley soils	41721 (9.8%)	125456 (29.4%)	167177	8.1
Total	62900 (14.8%)	363118 (85.2%)	426018	20.5



Map 2. The Soils of Wales Map (2019) developed through the Capability, Suitability & Climate Programme (Report CSCP02) classifies soils by soil series. This derived map represents soils by ALC textual categories. Deep peats and shallow peaty (Pty) soils are differentiated. This product attempts to identify specific soil series and smaller areas of peat as opposed to soil associations where peats may occur.

Current land use

Grassland currently occupies (65%) of the land area of organo-mineral soils in Wales, most of which is located in upland areas (222,119 ha) compared to lowland areas (44,623 ha) (Table 4). The categories for upland and lowland are based on Environmental Zones (EZ8 Lowlands of Wales and EZ9 Uplands of Wales) which are derived from ITE Land Classes (Bunce *et al.*, 1996). Grassland on organo-mineral soils

constitutes around 20% of the total area of grassland in Wales. However Welsh organo-mineral soils also support a range of other land use types (Bol *et al.*, 2011; Evans *et al.*, 2015) (Table 4).

Currently forestry occupies 16 % (66,972 ha) of the total area of organo-mineral soil in Wales. Most of the forested land underlain by organo-mineral soil (79%, 53,223 ha) is coniferous forest in upland areas with a much smaller area of coniferous forestry on organo-mineral soils found in lowland areas (5%, 3463 ha). Broadleaf forestry (14 % of total forested area on organo-mineral soil) occupies a similar area of land in the Welsh uplands (7% of total forested area on organo-mineral soil, 4892 ha) and lowlands (7% of total forested area on organo-mineral soil, 4895 ha).

Table 4. Area (ha) of organo-mineral soil types under different land use within Wales, adapted from Bol *et al.*, (2011). Organo-mineral soil types: (1) well drained; (2) podzols; (3) gleys.

Environmental Zone	Lowland Wales			Upland Wales			Total area (ha)	% of total organo-mineral area
	Soil type			Soil type				
	1	2	3	1	2	3		
Cropland								
Arable cereals	0	10	185	6	48	103		
Horticulture	1295	117	1369	259	522	911		
Non-annual	0	0	0	0	0	0		
Total	*1295*	*127*	*1554*	*265*	*570*	*1015*	*4826*	*1.1%*
Grassland								
Improved grassland	8514	1079	17665	5170	3576	11272		
Set aside	0	0	15	6	0	2		
Rough grass	1147	380	6057	3987	14802	37352		
Calcareous grass	462	42	1641	853	2603	2898		
Acid grass	2153	1385	3393	20676	81842	27669		
Bracken	183	156	349	1031	4186	4194		
Total	*12460*	*3042*	*29121*	*31722*	*107009*	*83388*	*266742*	*65.1%*
Heath								
Dense dwarf shrub heath	326	109	447	7399	16344	5394		
Open dwarf shrub heath	157	137	341	7560	14579	12782		
Total	*483*	*246*	*788*	*14958*	*30922*	*18176*	*65573*	*16%*
Forest								
Broadleaved	660	184	4051	1183	981	2728		
Coniferous	621	784	2058	4443	33794	14986		
Total	1281	968	6610	5625	34775	17713	66972	*16.3%*
Wetland								
Fen, marsh and swamp	0	5	66	109	284	251		
Bog	4	0	4	126	2690	230		
Standing/inland water	41	35	13	385	359	100		
Total	*46*	*40*	*82*	*620*	*3332*	*581*	*4701*	*1.1%*
Coastal								
Saltmarsh	49	6	63	25	24	1		
Supra-littoral rock	0	0	0	0	0	0		
Supra-littoral sediment	50	0	66	0	0	3		
Littoral rock	10	0	1	0	0	0		
Littoral sediment	52	0	41	0	0	2		
Total	*162*	*6*	*171*	*25*	*24*	*6*	*394*	*0.1%*

Organo-mineral soils are an important soil type for Welsh conservation areas with 23% of the organo-mineral soil area of Wales designated as land supporting Sites of Special Scientific Interest (SSSI) (Table 5) (Bol *et al.*, 2011). Furthermore, between 40-45% of the total area of Welsh Sites of Special Scientific Interest (SSSI), Special Protection Areas (SPA) and Special Areas of Conservation (SAC) are on organo-mineral soils (Bol *et al.*, 2011). A detailed analysis of the classification of Welsh organo-mineral soils into different land use categories is provided by Bol *et al.*, (2011). Land designations limit the range of management options which may be applied and may prevent certain areas from being considered as suitable for conversion to woodland/tree planting activities. A large proportion (80%) of the Welsh land area is designated as less favoured area (LFA) land. Over 97% (415,754 ha) of the total area of organo-mineral soils in Wales is classed as LFA land and organo-mineral soils constitute 25% of the total Welsh LFA area (Table 5).

Table 5. Areas of organo-mineral soils in Wales included within designated areas. Adapted from Bol *et al.*, (2011). Designation classes; SAC: Special Areas of Conservation; SPA: Special Protection Areas; SSSI: Site of Special Scientific Interest; NNR: National Nature Reserve; ESA: Environmental stewardship agreements; LFA: Less Favoured Area. The final column shows the proportion organo-mineral under each land designation out of the total land area under each designation on all soil types in Wales.

Wales Designation	Area (ha) 1 (well drained)	Area (ha) 2 (podzols)	Area (ha) 3 (Gley Soils)	Area (ha) All Categories	Total area (ha) designated (all soil types)	Organo-mineral proportion designated area (%)
SAC	19242	26632	10918	56792	138902	41
SPA	5504	23907	4266	33677	81319	41
SSSI	25018	49781	22415	97214	217306	45
NNR	2950	2504	1350	6804	21260	32
ESA	1795	12337	4127	18259	80302	23
LFA (total)	68767	186074	160913	415754	1634970	25
LFA (disadvantaged)	6733	23	14997	21753	473938	5
LFA (severely disadvantaged)	62034	186051	145916	394001	1161032	34

Carbon storage

Estimates of the carbon stock held by Welsh organo-mineral soils vary due to differences in the definition of organo-mineral soil types and method of classifying soil area. Based on data from Emmett *et al.*, (2010), Bol *et al*., (2011) estimated that organo-mineral soils hold approximately 41 Tg[3] (25%) of the 159 Tg[4] of soil carbon in Wales at 0-15 cm depth although this was based on a relatively small number (48) of sampling locations. In contrast the ECOSSE project estimated the total carbon stock of Welsh organo-mineral soils to be 74.5 Tg[5] (Scottish Executive, 2007). The differences in carbon stocks estimated by the two studies partly reflect differences in the definition of organo-mineral soil used and thus the estimated organo-mineral soil land area. The ECOSSE project used a more restricted definition of organo-mineral soil types which excluded the following soil sub-groups: humic rendzinas, humic brown podzolic soils, typical gley podzols, stagnogley podzols, pelo-alluvial gley soils, typical humic alluvial gley soils and typical humic sandy gley soils. In contrast these groups were included in the organo-mineral soil definition by Bol *et al.*, 2011. However both projects estimated that over 50% of

[3] 41 million tonnes of soil carbon
[4] 159 million tonnes of soil carbon
[5] 74.5 million tonnes of soil carbon

the total carbon stock of Welsh organo-mineral soils is stored at a depth of 0-15 cm (Scottish Executive, 2007; Bol *et al*., 2011) reflecting the variability of the depth of the peat layer (i.e.<40 cm depth) across the different soil types. This is important when considering vulnerability of soil carbon to land use change as shallow carbon stocks are more vulnerable to environmental and land management/land use pressure (Bol *et al.*, 2011).

The recent BioSoil survey by Vanguelova *et al.*, (2013) which used 167 sampling locations across the UK (26 in Wales) produced estimates of the amount of carbon stored in UK forest soils. Soil carbon was estimated to a depth of 80 cm using bulk density measures at four depths per sampling point making this study more reliable than previous estimates. For the whole of the UK, organo-mineral soils (peaty gleys, peat podzols and peaty rankers) were estimated to hold 321 t C/ha, compared to peat soils (448 t C/ha). Notably, the carbon content of organo-mineral soils was estimated to be double that of mineral soils (133-155 t C/ha) (Table 6).

Table 6. Estimated total soil carbon of UK forest soils to a depth of 80 cm from (Vanguelova *et al.*, 2013) of deep peat, organo-mineral and mineral soils.

Soil Class	Estimated tonnes CO_2 per ha	Estimated tonnes C per ha
Deep peat	1644	448
Organo-mineral	1174	321
Mineral	487- 570	133 -155

Climate projections for Wales and potential implications for woodlands

Climatic changes predicted for Wales are expected to affect woodlands in a range of ways (Table 7). A warmer climate is likely to increase the length of the growing season and capacity for growth and productivity. For example, warmer summer temperatures have been predicted to increase productivity of Sitka Spruce by 2-4 m^3 ha^{-1} yr^{-1} nationally provided that other requirements such as water and nitrogen are not limited (Ray, 2008). However, increases in frequency and severity of weather stress such as summer drought and fires, intense winter rainfall events and storm damage may pose an increased threat to woodlands (Welsh Government, 2018). Additionally, it is likely that changes in climate will affect the prevalence and distribution of woodland pests and disease.

Table 7. Projected climatic impacts and consequences for Welsh woodlands. Adapted from (Ray, 2008; Welsh Government, 2010).

Climatic variable	Projection	Potential impact upon Welsh woodlands.
Temperature	Summers will become warmer and winters milder with fewer frost days.	Longer growing season, growth rate of many common species will increase. Changes in provenance of some species e.g. Douglas fir. Extension of planting season for some trees, increasing weed competition. Change in pest and pathogen distribution and incidence.
Rainfall distribution	Drier summers in the east and south, wetter winters.	Water stress to trees, change in tree species suitability and productivity.
High intensity rainfall	Increased frequency particularly during winter leading to greatly increased risk of flooding and land slips and risk of soil erosion.	Tree stress, increased risk of pests and disease. Changes in tree species suitability and productivity. Increased risk of water erosion and sedimentation of water courses.

Drought	Increased frequency of drought, especially in south and fires in summer.	Tree drought stress and potentially increased risk of pests and disease. Change in tree species suitability and productivity. Limitations to planting season of bare root stock, reduced establishment success of young trees with less developed root systems.
Wind speed	More frequent strong winds.	Storm damage and limitations for some silviculture systems in exposed locations.

Future changes in climate and weather patterns are also important when considering the effects of forestry activities upon organo-mineral soils. There is much uncertainty regarding the impact of climatic changes on soil properties and functions however potential impact include:

- Extreme weather events such as heavy rainfall during early stages of woodland creation may increase susceptibility of soils to erosion particularly if planting occurs on steeply sloped land in upland areas.
- Summer drought may increase hydrophobicity of soils increasing susceptibility to erosion. Greater surface runoff during rainfall events may increase both soil erosion and flood risk.

Despite large uncertainties in predicting the impact of climatic change on organo-mineral soils, a review of this topic by (Bol *et al.*, 2011) concluded that it was likely that predicted increases in temperature and changes in the seasonality and magnitude of rainfall events would impact soil properties including carbon storage, soil structural stability and erosion susceptibility.

Susceptibility of soils to erosion with a changing climate may be exacerbated by land use (Boardman *et al.*, 1990). Bol *et al.*, (2011) suggested that soils under intensive agricultural management may be at increased risk of structural degradation under future climate scenarios. For example, more winter rainfall may reduce the period for which soil conditions are suitable to access agricultural land without causing compaction and structural degradation. This is also relevant to the management of soils for forestry where wetter soils in winter may limit the time that machinery can access land without causing structural damage. This could potentially lead to increases in GHG emission, soil erosion and degradation of soil and water quality.

The Welsh Government Woodlands for Wales Strategy (2018) has emphasised the importance of considering suitability of trees for future climatic conditions when creating new woodlands. Diversifying the species composition of new Welsh woodlands was proposed as one strategy for improving resilience to climatic change (Welsh Government, 2018). In suitable locations, woodland creation may help mitigate impacts of climate change upon other ecosystem services by increasing landscape resilience. For example strategically placed woodlands may help to alleviate downstream flooding (Thomas & Nisbet, 2007) whilst on farms trees may improve livestock performance during periods of climatic stress by providing shelter and shade (Woodland Trust, 2012).

The most recent UKCP18 predictions of how the UK's climate will change under different emission scenarios show changes to a range of climatic conditions across Wales (Lowe *et al.*, 2018). Cranfield University (2019) has applied the Agricultural Land Classification (ALC) climate interpolation routine to the UKCP18 models of Lowe *et al.*, (2018) to map changes in average annual rainfall (**Figure 3**), average summer rainfall (**Figure 4**), median accumulated temperature above 0°C January to June (**Figure 5**), median duration of field capacity days (**Figure 6**) under low, medium and high emission scenarios and for three time periods; 2020, 2050 and 2080. The main climatic changes predicted to effect upland areas where organo-mineral soils are predominantly located in Wales are summarised below:

- Annual rainfall: Average annual rainfall (**Figure 3**) shows little change for any time period/emission scenario with in general most areas receiving 1201-4500 mm rain annually. However average summer rainfall (**Figure 4**) shows reductions over time particularly in the south and east of Wales

for all emission scenarios with the area of land receiving >800 mm of average summer rainfall, considerably reduced.

- Temperature: Under all emission scenarios median temperature (Figure 5) is increased with the level of increase greater with higher emission scenarios. Greatest increases are shown for the high emission scenario.
- Field capacity days: Median field capacity days (Figure 6) for most of Wales are classed as 223-279 days or >280 days under all scenarios. Particularly in south east Wales there could be a reduction in field capacity days in 2050 and 2080 compared to 2020, particularly in the high emissions scenarios due to reduced summer rainfall.

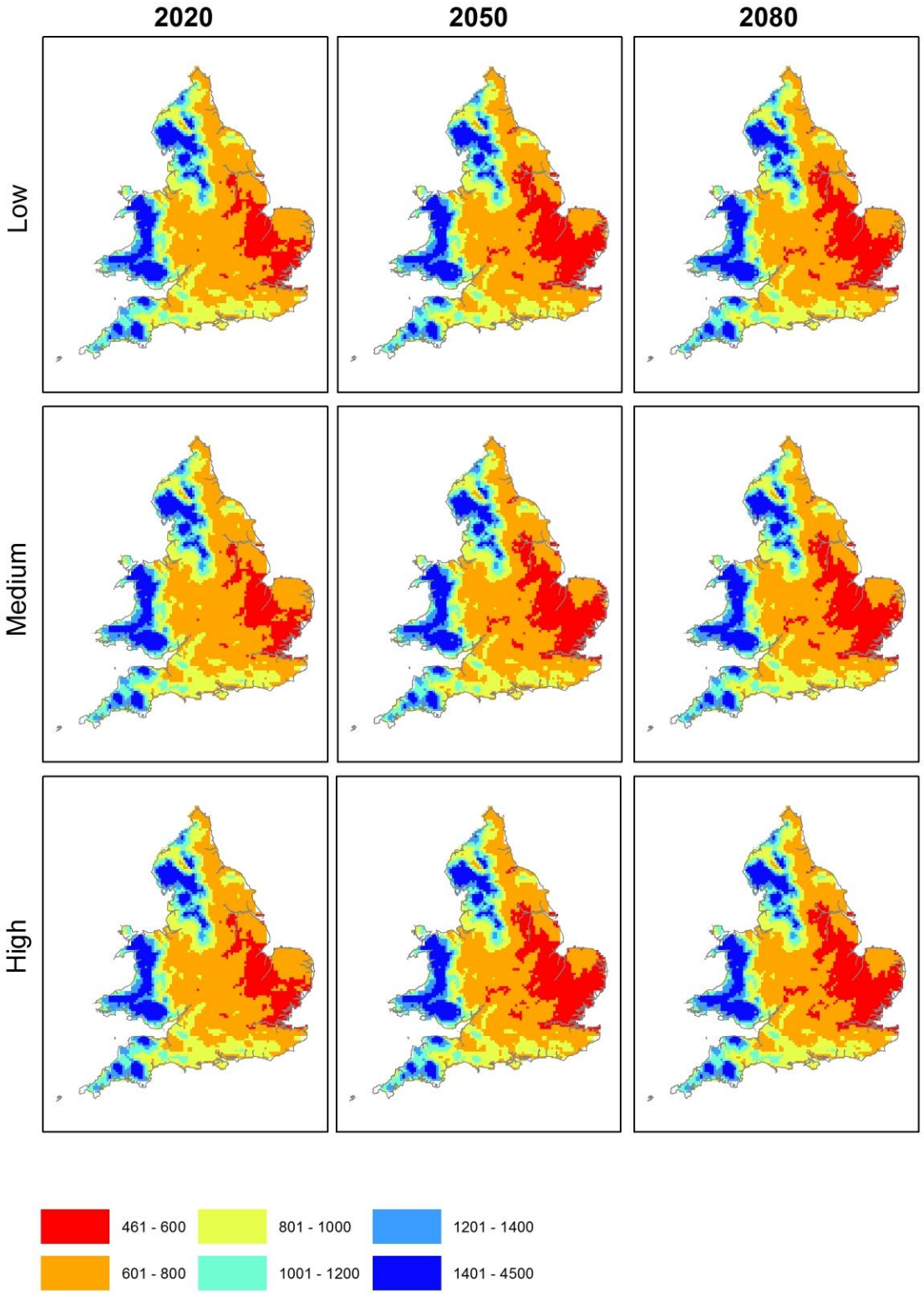


Figure 3. Average annual rainfall (mm) for 2020, 2050 and 2080 under UKCP18 low medium and high carbon emission scenarios.

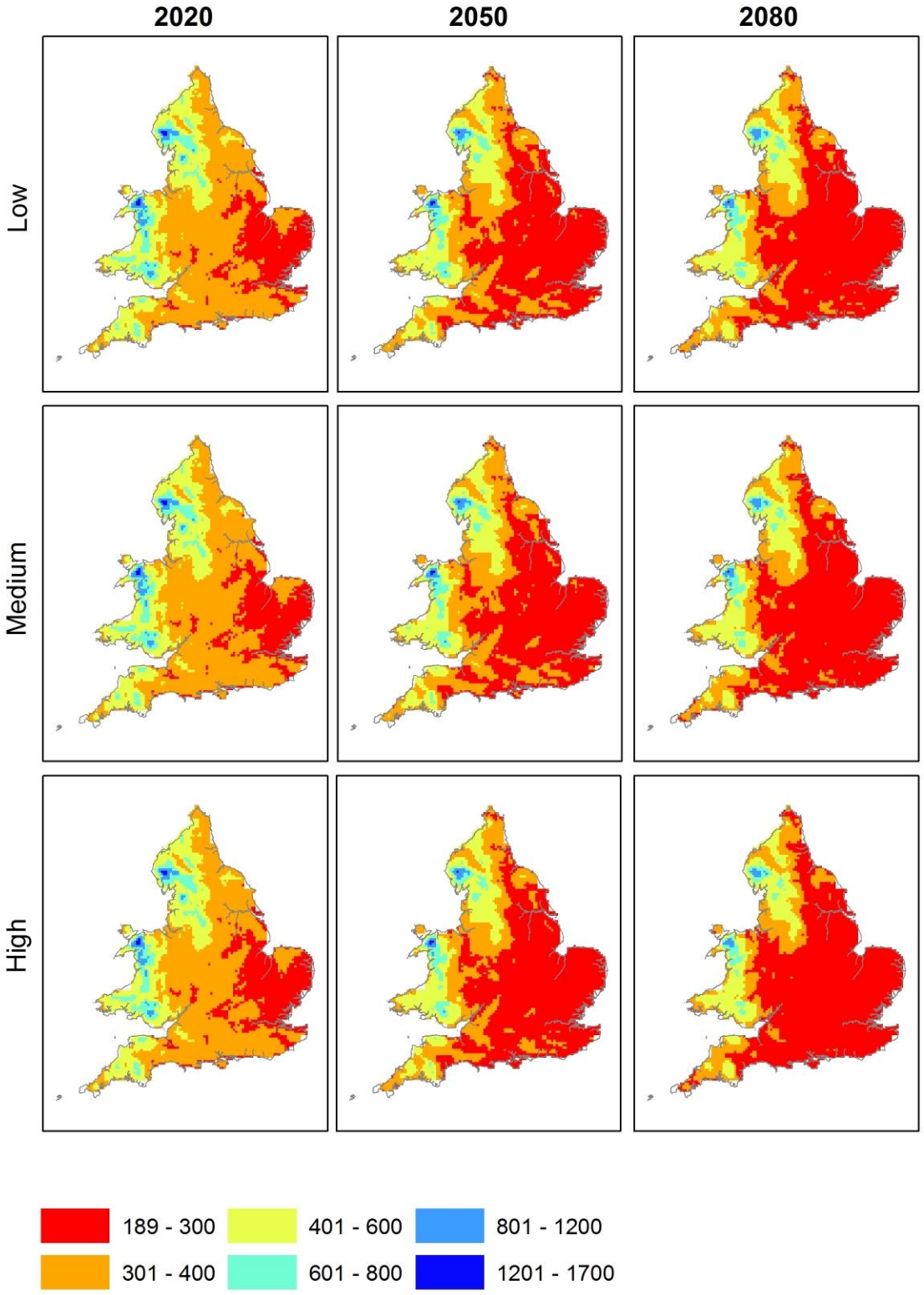


Figure 4. Average summer rainfall (mm) for 2020, 2050 and 2080 under UKCP18 low, medium and high carbon emission scenarios.

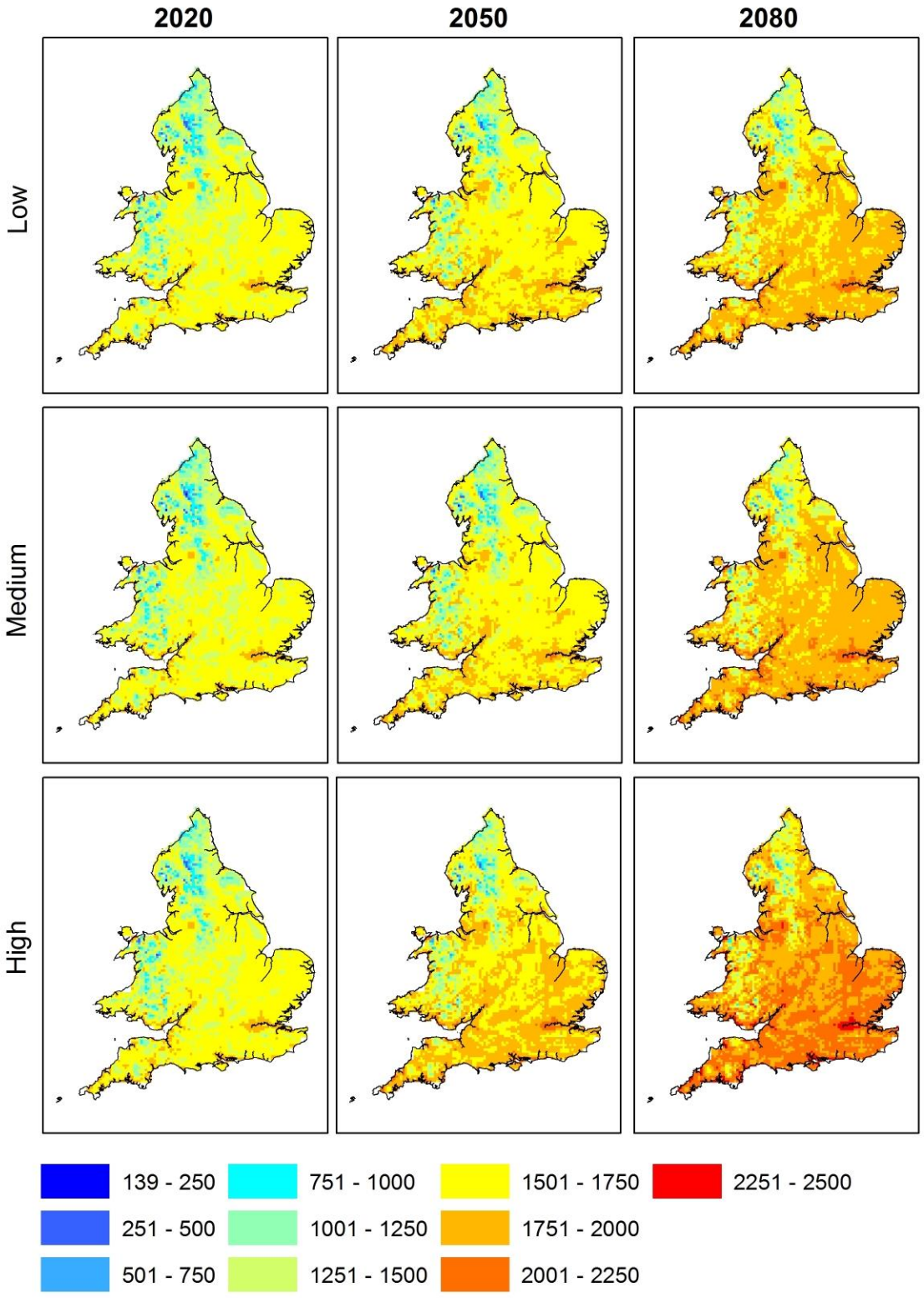


Figure 5. Median accumulated temperature above 0°C from January to June for 2020, 2050 and 2080 under UKCP18 low, medium and high scenarios.

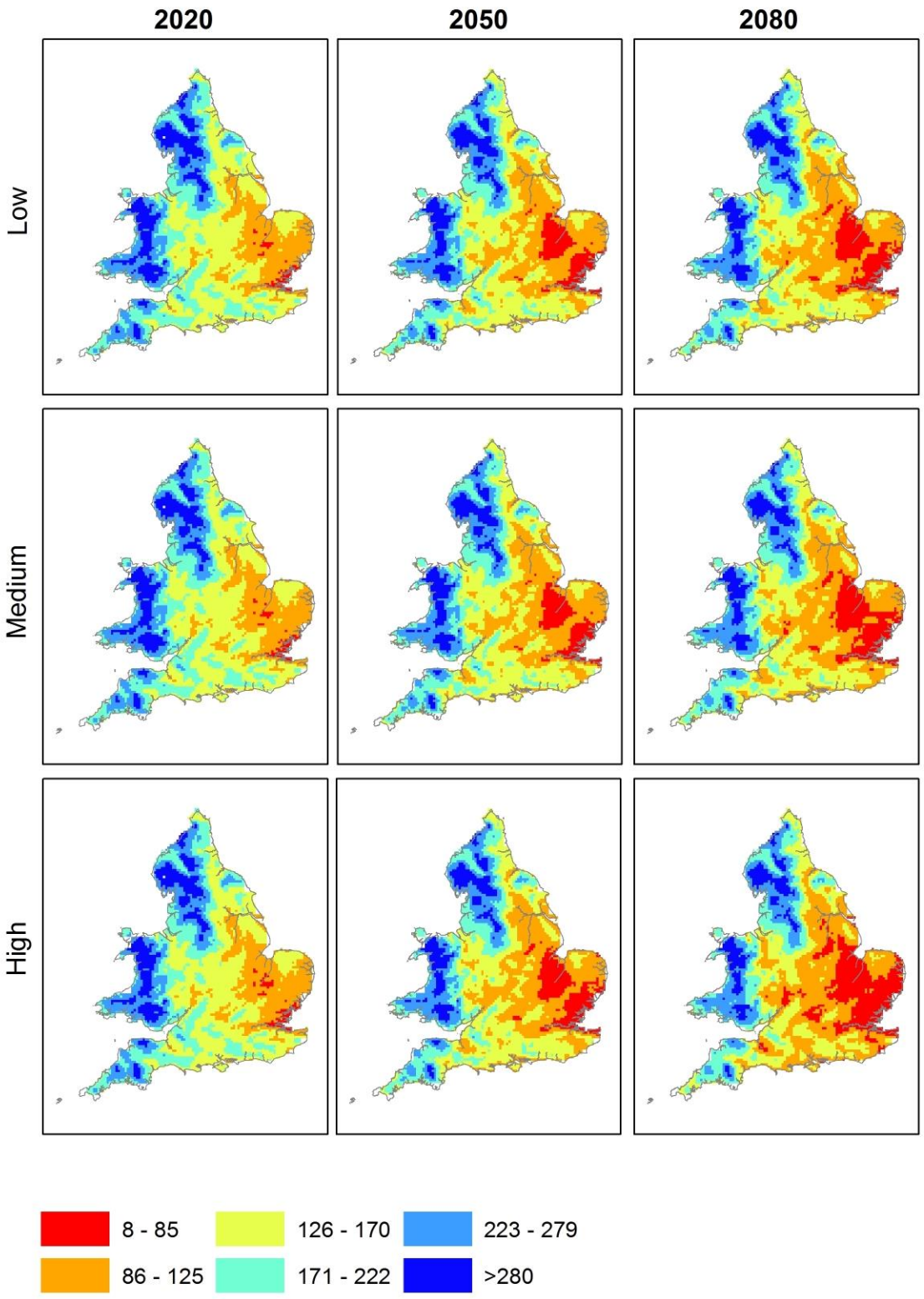


Figure 6. Median duration of field capacity (days) for 2020, 2050 and 2080 under UKCP18 low, medium and high carbon emission scenarios.

Impact of forest management upon soils

The impact of forestry upon soils is strongly dependent upon forest management with soil disturbance during planting and harvesting particularly likely to impact upon soil properties. Some of the most important processes which determine impacts of forest management upon soils and soil carbon are soil erosion, compaction, nutrient removal, and soil water status (Worrell & Hampson, 1997). In general it is thought that the carbon benefits from woodland creation are the greatest where soil is disturbed as little as possible by woodland establishment and management practices (West, 2011). For this reason, the Woodland Carbon Code recommends that where necessary, pre-planting ground preparation practices are selected which have the minimum soil disturbance required for successful tree establishment (West, 2011). Current guidance from the forestry industry also acknowledges the threat that soil disturbance poses to soil carbon and recommends minimising soil disturbance by forestry activities where possible to protect soil carbon stocks (Forestry Commission, 2017).

A detailed assessment of the carbon and greenhouse gas emissions associated with all aspects of forestry from ground preparation to management and harvesting is provided by Morison *et al.*, (2012) and shows the importance of considering the carbon implications of management decisions to achieve net carbon sequestration benefits with woodland creation. Currently Environmental Impact Assessments for forestry do not consider site specific soil carbon balances and forestry carbon balances generally classify organo-mineral soils as either peat or mineral soils which can lead to over or under representation of carbon content respectively.

Forest establishment

Organo-mineral soils often require management prior to afforestation to provide suitable conditions for tree growth particularly for areas with high soil moisture. Ground preparation may include the installation of drainage systems to reduce the water table, ploughing to increase soil aeration and herbicide application to remove vegetation cover. Both drainage and aeration increase rate of organic matter oxidation and carbon loss both atmospheric (as CO_2) and to groundwater as dissolved organic carbon (DOC) and particulate organic carbon (POC) (Morison *et al.*, 2012). It is generally accepted that loss of soil carbon increases with the level of soil disturbance (West, 2011; Morison *et al*., 2012). For organo-mineral soils in the UK, increased ground disturbance prior to or during forest planting may increase the loss of soil carbon due to accelerated rates of decomposition (Scottish Executive, 2007).

Ground preparation for large scale afforestation usually involves operations using heavy machinery. Soil preparation treatments differ in the amount and extent of soil disturbance. Estimates of the proportion of the site area disturbed as reported by Worrell, (1996) are: ploughing (44-66% soil area affected) > dolloping (soil inversion) (29-31% soil area affected), > disc trench scarifying (20-32% soil area affected), > disc patch scarifying (14% soil area affected), > hand turfing (4-7% soil area affected), > hand screefing (negligible soil area affected). The UK Forestry Standard recommends that to minimise impacts on soils the least disruptive method required to ensure effective tree establishment should be selected (Forestry Commission, 2017).

Manual planting methods generally cause lower levels of soil disturbance than mechanised planting. On grassland, manual planting has the advantage of allowing grass cover to be maintained between the tress which may reduce initial SOC loss and leaching (Thuille & Schulze, 2006). The three main manual planting methods described by the Woodland Trust, (2020) are pit planting, slit planting and T-notch planting (Table 8). The techniques differ in terms of time requirements and the soil conditions to which they are suited. The Woodland Trust suggests pit planting to be the most preferable method because despite being more time consuming than other methods, pit planting provides better contact of tree roots with soil which can help to alleviate drought stress in water limited areas (Woodland Trust, 2020a). When pit planting on grassland, the section of turf removed from the planting hole can be placed in the bottom of the pit to provide extra nutrients for the sapling (Woodland Trust, 2020a).

Table 8. Summary of manual planting methods. Adapted from Woodland Trust, (2020a)

Type	Description	Advantages	Disadvantages
Pit planting	A hole is slightly wider and deeper than the sapling root size must be dug. The tree is positioned in the hole and the soil is replaced.	Ensures good contact between tree roots and soil. Is suitable for all ground types including in drought prone areas.	Planting can be difficult in stony ground.
Slit planting	A slit is made in the soil using a spade. The slit is opened using a spade and the tree is placed inside with the root plug about 2 cm beneath the ground level. The spade is removed and the soil is pushed back around the tree.	Simple method suitable for planting in bare soil and grass.	Not suitable for drought prone areas.
T-notch planting	A spade is inserted into the ground. The spade is removed and inserted again to add a further slit at a right angle to original the slit and creating a 'T' shape. The spade is inserted back into the original cut and lever up to part the turf. The tree root plug is placed between the turf sections. The spade is removed, and the tree and turf are lowed back into the ground. The soil is firmed down around the tree base.	Quick method suitable for planting into grass, suitable for areas susceptible to drought.	Not suitable for planting in bare soil, not recommended for sites with clay soils.

Kerr & Williams, (1999) detail the considerations and processes involved in establishing a new woodland drawing on experiences from the creation of the National Forest which saw 8.9 million trees planted in the 1990's. The report highlighted important pre-planting considerations such as site condition and assessment, species choice, weed control and tree protection. Kerr & Williams (1999) suggest tree planting densities of 2500 trees per hectare with 2 m spacing, similarly the Woodland Trust (2020) recommends 2 metre spacing between trees although 1-5 m spacing may be used dependent upon woodland design. The Woodland Trust recommends planting trees as saplings between 1 and 2 years old when dormant in the winter months.

Natural regeneration whereby trees and eventually woodlands regenerate from the natural dispersal of seeds could also be used to create of new areas of woodland. This method relies on trees growing from a naturally dispersed seed bank and therefore is most suitable when used for expanding an established woodland or for connecting existing woodland sites or to accompany low density woodland planting (Forestry Commission, 2017; Woodland Trust, 2020b). The effect of natural regeneration on organo-mineral soil carbon stocks has not been measured however it may have less impact on soil carbon stocks that more disruptive planting methods as no soil disturbance is necessary (Lewis *et al.*,

2019). The role of natural regeneration for improving woodland resilience to climate change by creating woodland that is diverse in terms of both species diversity and genetic diversity has been recently recognised by the forestry industry (Forestry Commission, 2017). Improving genetic diversity of woodlands trees provides potentially greater capability for adaptation to climate change and threats from pests and pathogens. Naturally regenerated woodland contains a greater range of habitat structures and is thus able to support a wider range of wildlife (Woodland Trust, 2020b).

Harvesting

Harvesting methods contribute to carbon and greenhouse gas emissions from woodlands and a comprehensive assessment of this has been provided by (Morison *et al.*, 2012). Overall it is clear that for climate change mitigation and carbon sequestration purposes rotation and harvesting practices must be carefully considered. With regard to improving the sequestration potential of plantation, the following practices may be beneficial: (i) harvesting less frequently, (ii) using different species (species with higher carbon sequestration rates) (iii) converting timber into longer lived products. However these could all lower yields/ economic gains. From 2010-2019 Wales produced on average annually 1,296 thousand green tonnes[6] of softwood, 28 thousand green tonnes of hardwood. Of the softwood processed at large Welsh sawmills[7] in 2018 (most recent available data), 20% was used for construction, 27% fencing, 41% packaging and pallets and 12% other purposes including woodchip, bark and sawdust (Forest Research, 2019b).

Impact of woodland creation upon organo-mineral soil carbon stocks.

Forest carbon budgets

The overall carbon balance of the forestry system depends upon the net balance of the carbon stocks which are trees, debris, litter and soil (in forest stocks) and the harvested wood product and associated potential GHG emissions abatement of the product use (out forest stocks). The overall carbon budget of woodland and the carbon sequestration benefit of afforestation is determined by inputs and outputs of these stocks as summarised in Figure 7. The fate of harvested wood products (not included in Figure 7) also influences the net GHG emission balance as if accumulated (e.g. timber used in building), the wood product remains a carbon stock, whilst use as fuel or other short term purpose (e.g. paper) results in more rapid carbon release. If used as fuel, the fuel source (e.g. fossil fuel) that is replaced should also be considered due to potential overall reduction in GHG emissions. Detailed assessment of complete forest carbon budgets are provided elsewhere for example (Morison *et al.*, 2012; Woodland Carbon Code, 2018). Whilst soil carbon only comprises one component of the carbon stocks within temperate forest, forest soils generally hold substantially more carbon than any other stock within the system (Janzen, 2004). For example, it has been estimated that Welsh forests have a soil C stock (to 1 m) of 201 $MtCO_2$ whilst the C stock of tree biomass and litter/deadwood is 72 $MtCO_2$ and 17 $MtCO_2$ respectively (Morison *et al.*, 2012). Therefore, if the aim of woodland creation is to provide a net carbon sequestration benefit, considering the impact of forestry upon soil carbon is important.

[6] Undried tree biomass. One green tonne is approximately equivalent to 0.98 m^3 underbark softwood/0.88 m^3 underbark hardwood/1.22 m^3 overbark standing softwood/1.11 m^3 overbark standing hardwood.

[7] Sawmills producing at least 25,000 m^3 sawnwood (hardwood and softwood).

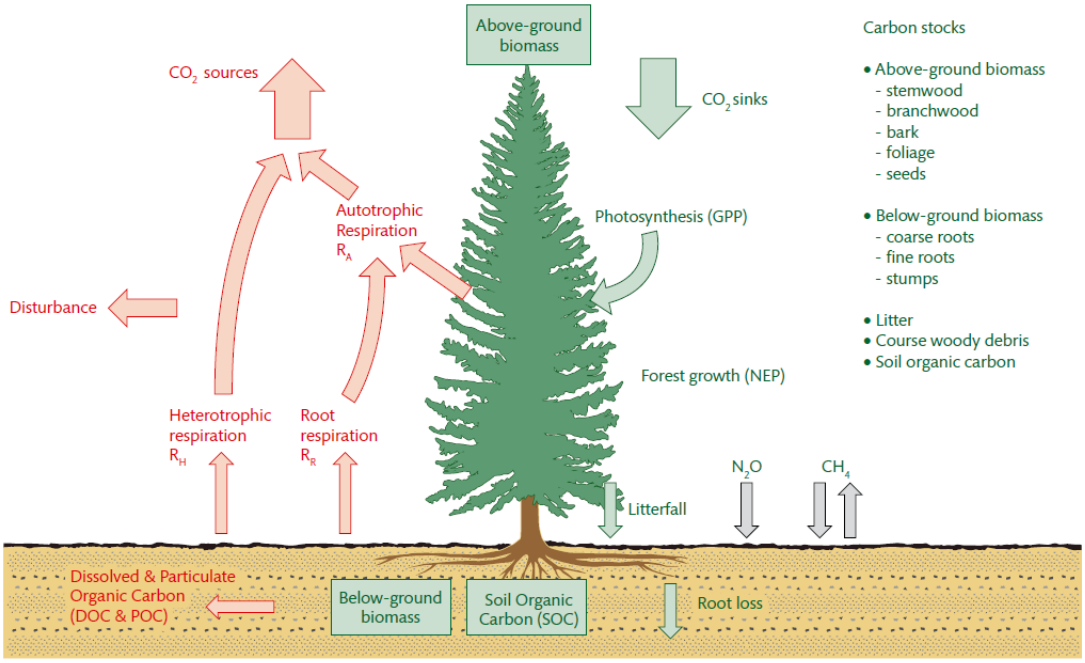


Figure 7. Processes and fluxes maintaining or modifying forest carbon stocks and fluxes of other GHGs (CH_4 and N_2O) from (Morison *et al.*, 2012). Green arrows indicate carbon flux into the forest and between various carbon stocks, red arrows indicate carbon fluxes out of the forest. The two main sources of soil respiration are heterotrophic (soil micro-organisms decomposing organic material) and autotrophic (respiration by live roots and associated mycorrhizal fungi).

Soil carbon stocks

The soil organic carbon (SOC) stock is determined by the balance of carbon inputs and losses. Total carbon allocation belowground is the net input from roots and mycorrhiza taking into account production, respiration and exudation and addition to the soil surface from organic material such as leaf litter. Main soil carbon losses arise from microbial oxidation of organic matter to carbon dioxide (CO_2), dissolved organic carbon (DOC) or particulate organic carbon (POC). Soil carbon may be lost more rapidly than the rate at which it can accumulate (Freibauer *et al.*, 2004). The overall impact of woodland creation on organo-mineral soil carbon stocks is determined by:

- **Soil characteristics** such as the soil depth, organic carbon content, clay content, bulk density and properties such as pH and cation exchange capacity. The previous land use and management prior to afforestation/reforestation will also influence the SOC stock at planting and the net impact of afforestation on soil carbon stocks.
- **Management practices** particularly those which cause physical soil disturbance such as drainage, tree planting and harvesting may result in carbon losses by increasing rates of mineralisation and leaching of DOC and POC. The level of soil disturbance and subsequent carbon loss will depend on the methods used for planting, management and harvesting. The period of forest growth or rotation length will also influence the overall effect of forestry on soil carbon stocks as this will determine the frequency of soil disturbance from harvesting, planting and management such as thinning. For example whilst Sitka spruce is typically grown in 35-40 year rotations (Moore, 2011) rotation lengths vary from around 20 to 60 years.
- **The tree species** that are planted and the tree physiology including rates of CO_2 uptake, growth, root development and turnover, and the quantity and quality of litter deposited will determine the soil carbon inputs. Tree species will also influence soil carbon cycling processes and decomposition rates indirectly for example by affecting soil properties such as soil pH (through litter inputs) and by altering aspects of the soil microclimate such as moisture content and light levels due to

differences in canopy density and development. Species selection is therefore extremely important for determining effects of afforestation/reforestation upon soil carbon stocks.

- Environmental conditions for example temperature and rainfall, influence soil carbon stocks directly by affecting the rate of processes which determine carbon inputs (e.g. tree productivity and carbon assimilation) and losses (e.g. microbial activity and soil respiration). Environmental conditions will also indirectly influence carbon stocks by affecting the above detailed aspects of forestry systems (soil properties, the impact of management practices and tree physiological processes). However, changes in environmental conditions and how these will impact organo-mineral soil carbon stocks in the long term are particularly difficult to predict.

Effect of afforestation upon soil carbon

To date most studies examining the impact of afforestation on organo-mineral soils have been studies of Sitka spruce (*Picea sitchensis*) plantations in northern England and Scotland over one or two rotations. Studies to date have been short term (first forest rotation) or medium term (two rotations, up to 100 years) however there are currently no long-term studies (over 100 years) of carbon change resulting from the afforestation of organo-mineral soils. There is also no evidence in the literature of studies investigating changes in organo-mineral soil carbon stocks under broadleaf forest.

It is clear from studies to date, that planting and harvesting of coniferous forest can cause substantial carbon loss from organo-mineral soils and that in the short-term (one forest rotation) the soil carbon stock is usually reduced by afforestation. The most recent evidence from studies of commercial coniferous forestry have suggested that it is possible that when trees are planted using low soil disturbance methods, forestry using moderate and high productivity trees may result in little or is some cases no significant change to net carbon storage when the full cycle of forest growth is considered over two forest rotations (Vanguelova *et al.*, 2018). However, the long-term effects of forestry upon organo-mineral soil carbon stocks are uncertain. The available evidence examining the impacts of forestry upon organo-mineral soil carbon stocks are summarised below:

Peat layer carbon change

- Substantial losses in carbon from organo-mineral soils have been reported for the first 40 years following afforestation with coniferous plantation (Table 9) however there are no studies reporting whether similar losses occur with conversion of organo-mineral soil land use to other types of forestry.
- Vanguelova *et al.*, (2019) reported a significant loss of carbon at a rate of 1.8 t C ha^{-1} yr^{-1} from the peat layer of a peaty gley soil in northern England during the first 30 years of afforestation with Sitka spruce compared to adjacent moorland soils. At a similar forest site in northern England, Vanguelova *et al.*, (2018) report losses of 2.8 t C ha^{-1} yr^{-1} (measured to a depth of 50 cm) with 35 years of Sitka spruce afforestation. At this site soil carbon stocks to 50 cm depth were substantially reduced beneath plantation (160 t C ha^{-1}) compared to beneath adjacent heather moorland (215 t C ha^{-1}). Zerva *et al.*, (2005) reported even higher rates of carbon loss following 40 years of afforestation with Sitka spruce plantation at another peaty gley soil in northern England. In this case afforestation reduced total soil carbon (measured to a depth of 50 cm) to 140 ± 15 t C ha^{-1} compared to stocks of 274 ± 54 t C ha^{-1} measured in adjacent unplanted grassland. This equates to soil carbon loss at a rate of 3.3 t C ha^{-1} yr^{-1}.
- Lilly *et al.*, (2016) measured changes in soil carbon at 23 shallow peat sites across Scotland (with sampling on average 37.5 years after afforestation (with a range of 21 – 57 years). This study reported a mean loss of *c*. 80 t C ha^{-1} with first rotation spruce plantation on gley soils. However, a very large variation in carbon change with afforestation was reported between sites (+10 to -150 t C ha^{-1}) therefore the overall effect of afforestation upon soil C was not statistically significant. This large variation may reflect differences between the study sites particularly original land use (which included heather moorland, bog moorland rough grassland), length of rotation (21-57 years) and

differences in site characteristics (particularly latitude which ranged from North to South Scotland). Furthermore, the authors cautioned against site specific comparisons before and after afforestation due to low level of sample replication and the variability in thickness (particularly of the organic layers) at each site.

Table 9. Summary of studies of soil carbon change with Sitka spruce forestry on organo-mineral soils over the first forest rotation.

Soil type	Location	Length of first rotation (years)	C change with forestry	Original land use	C stock at end of rotation 1: forest	C stock at end of rotation 1: original land use	Reference
Peaty gley	Northern England	30	-1.8 t C ha^{-1} yr^{-1}	moorland	-	-	(Vanguelova *et al.*, 2019)
Peaty gley	Northern England	35	-2.8 t C ha^{-1} yr^{-1}	heather moorland	160 t C ha^{-1} (to 50 cm)	215 t C ha^{-1} (to 50 cm)	(Vanguelova *et al.*, 2018)
Peaty gley	Northern England	40	-3.3 t C ha^{-1} yr^{-1}	grassland	140 ± 15 t C ha^{-1} (to 50 cm)	274 ± 54 t C ha^{-1} (to 50 cm)	(Zerva *et al.*, 2005)

- In the studies of Zerva *et al.*, (2005) and Vanguelova *et al.*, (2018, 2019) ground preparation prior to planting involved ploughing and the creation of drainage ditches, activities which are known to breakup soil structure and soil aggregates which accelerates organic matter decomposition and the subsequent loss of carbon (Guo & Gifford, 2002). The reported high levels of carbon loss were attributed to leaching, oxidation and decomposition processes linked to the high level of soil disturbance at planting from these activities.
- Whilst the peat layer loses carbon initially (0- *c.*35 years) following afforestation, carbon stocks may then gradually begin to recover. (Vanguelova *et al.*, 2019) reported that despite initial carbon loss from the peat (H) layer during 0-30 years of the first rotation Sitka Spruce forestry (a loss of 94.3 t C ha^{-1} which equates to approximately 30% of the total C stock of the peat layer), the peat layer recovered 52.3 t C ha^{-1} over the second half of the rotation (30 - 60 years). It was suggested that this recovery was partly due to carbon from litter inputs and movement of carbon from the organic fermentation horizon down into the peat layer. The long term implications of this redistribution of carbon within the soil profile upon the biological functioning of organic layers is unclear and requires further research

Litter layer

- Afforestation of organo-mineral soils can lead to a gradual increase in carbon in litter and forest floor soil layers due the accumulation of leaf litter (Zerva *et al.*, 2005; Lilly *et al.*, 2016; Vanguelova *et al.*, 2019). Vanguelova *et al.*, (2019) reported that afforestation of peaty gleys in northern England resulted in the formation of a forest litter layer with an average litter layer carbon stock of 4.11 t C ha^{-1} with Sitka spruce afforestation whilst litter layers were absent from adjacent moorland. In comparison, Lilly *et al.*, (2016) reported that afforestation of shallow peats at sites across Scotland significantly increased the litter layer carbon stock by an average 0.53 t C ha^{-1} yr^{-1} (0.38-0.67, 95 per cent confidence interval) due to increased litter layer thickness.
- Zerva *et al.*, (2005) reported increases in the carbon stocks of litter layers in the second rotation of Sitka spruce forestry at 1.1, 1.1 and 1.7 t C ha^{-1} yr^{-1} for 12, 20 and 30-year-old second rotation stands respectively indicating an increase in litter accumulation with stand age. This is presumably due to the larger amount of leaf litter deposited by larger more mature trees.

Organic fermentation layer

- In addition to increases in litter layer carbon stocks, forest floor and organic fermentation layer carbon stocks may change with afforestation. Vanguelova *et al.*, (2019) reported both the thickness and carbon stock of the organic fermentation layer increased significantly over time with an average increase of 0.73 t C ha^{-1} yr^{-1} (ranging from 0.28-0.94 t C ha^{-1} yr^{-1}) when comparing Sitka spruce stands of different ages (up to two forest rotations and total of 100 years) within Kielder forest in northern England. Similar rates of increase in carbon by 0.54 t C ha^{-1} yr^{-1} for the top 15 cm of soil (but including the litter layer) have been reported by Chapman *et al.*, (2013) over 25 years of afforestation with coniferous woodland in Scotland. Similar results were reported by Lilly *et al*., (2016) who measured an average rate of carbon increase of 0.53 t C ha^{-1} yr^{-1} for the soil F horizon with 30-40 years of afforestation within Sitka spruce plantations across Scotland.
- Although effects on carbon storage in forest subsoils have been less studied, there is some evidence that afforestation on organo-mineral soil types has resulted in soil carbon moving deeper into the soil profile from upper peat layers and accumulating in underlying mineral soil (Swain *et al.*, 2010). Vanguelova *et al.*, (2019) reported increases in soil carbon concentration in the mineral A horizon of afforested peaty gley soils (0-20 cm depth) however carbon stock of the mineral layer did not change significantly.

Net effect upon carbon stocks over multiple rotations

- There is recent evidence that over multiple rotations, organo-mineral soil carbon stocks may at least partially recover carbon lost during the first rotation (Vanguelova *et al.*, 2018; Campbell & Robson, 2019). This processes of carbon recovery will be affected by the extent of physical soil disturbance during management, site specific factors such as nutrient status of soil and soil moisture and the physiology of the trees planted including tree productivity (Morison *et al.*, 2012).
- A recent study by Vanguelova *et al.*, (2019) quantified changes in soil carbon stocks to a depth of 70-100 cm for Sitka spruce plantation stands in northern England on peaty gley soils ranging in forestry ages up to 100 years (comprising of two forest rotations). Adjacent moorland control soils were compared with Sitka spruce stands with first rotation 30-year growth, first rotation 60-year growth, clearfell after 60 years afforested, and second rotation plantations after 5, 15, 25 and 40-60 years of growth. Across two rotations (with the maximum age being 100 years following afforestation) conifer afforestation had no significant change overall on the total carbon stock (in the top 70-100 cm soil). The average change over two rotations for the whole profile (forest floor (F), peat (H) and mineral (A)) was 0.14 t C ha^{-1} yr^{-1} (varying from 0.18 to 0.54 t C ha^{-1} yr^{-1}). Whilst carbon was lost initially lost from peat layers, carbon was gained in the litter and forest floor layers (organic fermentation horizon).
  - Similar recovery in soil carbon stocks over multiple rotations of Sitka spruce forestry on peaty gley soils in northern England was reported by Zerva *et al.*, (2005). This study reported that after two forest rotations, soil carbon stocks (to a depth of 50 cm) returned to levels similar to adjacent unplanted grassland at which point the annual rate of carbon accumulation was estimated to be 6.8 t C ha^{-1} yr^{-1}. The study compared unplanted grassland, 40-year old first rotation plantation, clear felled sites (sampled 18 months after clearfelling of first rotation plantation) and second rotation plantation after 12, 20 and 30 years of growth (Figure 8). Whilst initially soil carbon was reduced under the first rotation of forestry, by the second rotation soil carbon progressively increased from 147 ± 8 t C ha^{-1} after 12 years to 181 ± 17 t C ha^{-1} after 20 years and 249 ± 40 t C ha^{-1} by 30 years.
  - The impact of ending forest rotations and changing from forestry back to other land use on organo-mineral soils has not been quantified for soil carbon stocks. It is likely that the change in carbon stocks will depend upon the land use replacing forestry and interacting effects upon the soil environment which influence rates of carbon loss and accumulation. The evidence to date suggests that the recovery of soil carbon under second rotation coniferous forestry has been at least partly

due to the large amount of leaf litter that can be produced by productive forestry species such as Sitka spruce. Therefore, if forestry was stopped after the first rotation it is likely that soil carbon will take longer to recover with less productive vegetation types.

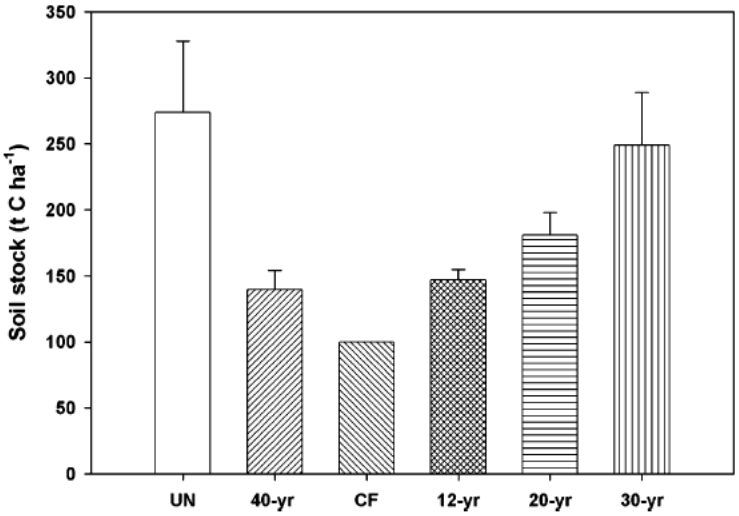


Figure 8. From Zerva *et al.*, (2005). Soil carbon stocks to 50 cm depth (t C ha^{-1}) comparing first and second rotation Sitka spruce plantation stands of different ages in northern England. Error bars show SE of mean. Treatments: UN (unplanted adjacent grassland), 40-yr (first rotation plantation with 40 years growth), CF (clear felled plantation measured 18 months after felling), 12-yr (second rotation plantation after 12 years growth), 20-yr (second rotation plantation after 20 years growth) and 30-yr (second rotation plantation after 30 years growth).

Forestry practices

Forestry practices which particularly affect organo-mineral soil carbon stocks are:

- Ground preparation: The extent of disturbance varies (soil disturbance: deep ploughing (35-50%), shallow ploughing (18-28%), no ploughing – and other preparation method (4-12%). It is generally accepted that the greater the ground disturbance the more carbon is lost however, more data is needed comparing the carbon and GHG balances caused by differences in soil preparation. The Forestry Commission has produced soil cultivation advice with the aim of reducing the negative impacts of planting upon soils including carbon loss (Forestry Commission, 2017). Ploughing is particularly disruptive and can cause loss of soil carbon, nitrous oxide (N_2O) and DOC.
- Whole tree harvesting including brash removal may be beneficial for increasing soil carbon stocks of organo-mineral soils as when brash is removed, less decomposition of soil organic matter occurs in peat soil layers. Comparison of whole tree harvesting with conventional harvesting (tree trunk removal), showed higher soil carbon stocks where whole tree harvesting had occurred. This was thought to be due to increased decomposition and mineralisation where conventional harvesting had been performed (Vanguelova *et al.*, 2010; Mäkiranta *et al.*, 2010). In a study comparing the effects of brash removal on soil carbon stocks of peaty gley soils under second rotation Sitka spruce stands located in Northern England 28 years after harvesting, Vanguelova *et al.*, (2010) found that carbon stocks within the peat horizon were significantly lower where brash was left on site (72 t C ha^{-1}) compared to where brash had been removed (108 t C ha^{-1}). This was attributed to increased carbon loss from mineralisation of brash and peat where brash was retained.
- Stump removal may result in significant loss of soil carbon especially upon shallow peat soils (Vanguelova *et al.*, 2017). This was demonstrated by a recent study in Bala, upland Wales on peaty gley soils which compared the soil carbon stocks of conventionally cleared and stump removed forest, 4 years after harvesting. The authors reported that peaty gley soils lost 220 t C ha^{-1} more

carbon to a depth of 80 cm (39% of the total estimated C stock) with stump removal compared to conventional harvesting (Vanguelova *et al.*, 2017). Stump removal is highly disruptive to soil structure and the physical soil disturbance leads to carbon loss as CO_2 from soil as well as by leaching of DOC (Collison *et al.*, 2015; Kaarakka *et al.*, 2018).

- Clearfelling has been shown to reduce organo-mineral soil carbon stocks due to physical soil disturbance. After clear felling CO_2 is released from decomposition of dead plant material (e.g. roots, surface litter). Changes in water table following clearfelling may also effect the GHG balance depending upon how dry the soil becomes. Zerva *et al.*, (2005) reported carbon stocks of 100 ± 13 t C ha^{-1} in peaty gley soils 18 months after clearfelling of first rotation Sitka spruce compared to 140 ± 15 t C ha^{-1} in 40-year old first rotation Sitka spruce stands. The decrease in soil carbon following clearfelling was thought to arise from soil disturbance through felling activities which included ploughing and mounding (inversion of 10-15% of the soil surface to a depth of 30 cm to create mounds of soil for new planting) which may increase loss of carbon through both oxidation and leaching loss in water (Vanguelova *et al.*, 2017), and also the lack of litter inputs following tree removal. Although the implications of the time period between clear-felling and restocking upon carbon losses has not been quantified it is likely that the longer the disturbed soil is left exposed the greater the potential for carbon loss through these processes. It has been suggested that changes to soil temperature after clearfelling may also increase rates of organic matter decomposition and affect other biological activity (Ballard, 2000).
- Drainage: A meta-analysis of studies investigating the impact of land management upon GHG flux in boreo-temperate peatland systems indicated a consistent trend of increased CO_2 and N_2O losses from drained peatland (Haddaway *et al.*, 2014). Studies of drained peatland soils have shown that as soil moisture is reduced by drainage, soil temperature may be increased. Increased CO_2 emissions from drained peat soils may be partly due to changes in microbial activity in response to warmer drier conditions (Mojeremane *et al.*, 2010).

Land use and soil carbon

Current understanding of the effects of conversion of different land use types on organo-mineral soils to forestry are summarised below:

Semi-natural upland grassland and improved grassland

- Assessing impact of conversion from semi-natural upland grassland is important as this is the land use of the majority (65%) of the organo-mineral soil area within Wales. Over 80% (*c.*220,000 ha) of these grasslands are semi-natural (unimproved rough or acidic) and over 90% are in upland regions.
- Temperate grasslands are a net carbon sink estimated to sequester 0.1-4 t CO_2e ha^{-1} yr^{-1} (Soussana *et al.*, 2004) however grassland management influences the carbon storage capacity. In particular soil disturbance and grazing intensity may influence grassland carbon storage with minimal disturbance and low grazing pressure generally optimal for increasing carbon sequestration (Bol *et al.*, 2011; Rollet & Williams, 2020).
- There is still uncertainty over how soil carbon changes with the conversion of semi-natural grassland to forestry on organo-mineral soils. It is generally assumed that under UK climatic conditions land use change will lead to reductions in the rate of soil carbon accumulation in the short term. In a study on a peaty gley site in Northern England, Zerva *et al.*, (2005) found that despite initial carbon losses, by 30 years into the second rotation of Sitka spruce forestry the soil carbon stock reached 249 ± 40 t C ha^{-1} whilst the carbon stock in the adjacent unplanted grassland was 274 ± 54 t C ha^{-1}.
- There is relatively little (<50,000 ha) improved grassland on Welsh organo-mineral soil. Bol *et al.*, (2011) assumed that the effect of conversion of this grassland type to forestry would be similar to that of semi-natural grassland.

Moorland and heathland

- Recent studies examining effects of afforestation of moorland upon soil carbon stocks report no change or small positive changes. Lilly *et al.*, (2016) found that afforestation of organo-mineral soils previously under moorland vegetation increased soil carbon stocks on average by 0.53 t C ha^{-1} yr^{-1} over 40 years however (Vanguelova *et al.*, 2019) found that after two rotation of Sitka spruce forestry, soil carbon under the plantation was not significantly different from that of the adjacent moorland.

Arable land

- Arable land accounts for only *c*. 1% of the organo-mineral soil area in Wales (Bol *et al.*, 2011).It is generally accepted that conversion of cropland to woodland would result in a significant increase in soil carbon due to increased carbon inputs from tree litter and decreased oxidation of organic matter. Forest vegetation is also able to store much larger amounts of carbon above ground in comparison to arable plant biomass (Liski *et al.*, 2002; Hooker & Compton, 2003).
- It is estimated that conversion of cropland to forestry could result in annual increases of 1.1-2.2 t CO_2e ha^{-1} yr^{-1} over the first 20-30 years following conversion (Soussana *et al.*, 2004; Johnston *et al.*, 2017). Conversion of arable land to forestry may result in the greatest increase in carbon stocks with organo-mineral soil land use change to forestry. However, there is a clear trade-off with the capacity for food production.
- Although sequestering less carbon than total forest conversion, integrating trees into farming systems using agroforestry systems can also provide benefit for carbon sequestration. Saunders *et al.*, (2017) estimated that over a 55-year period, net carbon uptake could be increased by between 55 - 107 t C ha^{-1} by the low-density planting of trees on grassland dependent upon the productivity of the site. Increasing tree cover on farms can provide additional benefits e.g. creating shelter and shade for livestock, reducing soil erosion, reduced nutrient leaching, fuel provision and supporting biodiversity (Woodland Trust, 2012; Saunders *et al.*, 2017). Improving tree cover on farms may also help to reduce diffuse water pollution of NO_3, P and sediment (Bol *et al.*, 2011).

Above ground biomass

- Land use change will also affect the amount of carbon stored in vegetation biomass. The vegetation carbon stock is estimated to be 30 t CO_2 ha^{-1} (8 t C ha^{-1}) for permanent grassland and 40 t CO_2 ha^{-1} (11 t C ha^{-1}) for heather moorland (Morison *et al.*, 2012) although these values will vary due to differences in environmental conditions and management practices. The carbon stock within forest vegetation biomass is likely to exceed these values at an early stage of afforestation. For example Morison *et al.*, (2012) estimated carbon stocks of 300 and 500 t CO_2 ha^{-1} for a 40-year old second rotation Sitka spruce plantation and a 70 to 80-year old deciduous oak plantation respectively. Although carbon accumulation rates vary between tree species, harvesting regimes and environmental conditions above ground carbon accumulation by forest biomass is substantially greater than for any other vegetation type (Liski *et al.*, 2002; Hooker & Compton, 2003).

Uncertainties

There are still many uncertainties regarding the impacts of afforestation upon organo-mineral soils. Key evidence gaps include:

- Organo-mineral soils of upland areas are often spatially heterogenous and changes in carbon stocks are relatively slow. Recent work by Vanguelova *et al.*, (2019) suggested that considering both the range of climatic conditions where organo-mineral soils occur within the UK and the range of organo-mineral land use types, it is currently not possible to be able to predict with certainty how afforestation and long term forestry management will impact soil carbon stocks in upland soils.
- Further work is needed to elucidate the effects of afforestation and forest management upon soil carbon stability. There is a knowledge gap regarding understanding of the different stabilisation

mechanisms of organic carbon within peat and mineral soil layers. There is evidence that forest management could influence the turnover of SOM in soils and thus effect carbon sequestration (Clarke *et al.*, 2015). Recent studies have suggested that practices such as ploughing and inversion of soils by mounding can cause translocation of carbon deeper into the soil profile (Swain *et al.*, 2010) however this requires further investigation.

- Further long-term studies investigating the impact of forestry on carbon stocks of organo-mineral soils are required as to date studies have principally focused on afforestation with coniferous tree species (mainly Sitka spruce) across one or two forest rotations (up to 100 years). Given the long time periods involved with forestry systems, one effective way of comparing long term changes in soil carbon with forestry is by using chronosequence studies whereby different aged stands and rotation lengths of forest within the same locality can be compared. This is important because long-term studies are required to fully understand the overall effect of forestry on soil carbon content. One limitation of the use of long term studies to date is that forestry practices have changed significantly over the last 40 years with an increasing focus on improving sustainability of soil management (Worrell & Hampson, 1997; Forestry Commission, 2017). This suggests that long term studies based on outdated management may not be relevant for understand impacts of modern forestry management upon soils.
- Further work is required to improve understanding of the impact of forestry on full GHG budgets (carbon dioxide, methane and nitrous oxide) as all of these gases are impacted by land use change. Not all current models investigating GHG emissions from forestry consider methane and nitrous oxide emissions.

Differences in carbon sequestration between hardwoods and softwoods

Carbon sequestration by woodlands is partly determined by the tree species planted and the physiology of the trees including differences in the rate of CO_2 uptake, growth rates and amount of litter deposited. An estimated 1,419 tonnes of CO_2e is sequestered by Welsh woodlands every year (Warren-Thomas & Henderson, 2017). In broadleaves, of which most in Britain are deciduous, 46% of wood mass is carbon compared to 42% in conifers (Milne & Brown, 1997). Conifer (softwood) species have a lower net carbon density than equivalent aged broad-leaved deciduous (hardwood) species (Morison *et al.*, 2012).

Hardwoods native to Wales

The following broadleaf species are native to Wales: alder (*Alnus glutinosa*); ash (*Fraxinus excelsior)*; aspen (*Populus tremula*); beech (*Fagus sylvatica)*; birch, silver (*Betula pendula)*; birch, downy (*Betula pubescens)*; cherry, bird (*Prunus padus)*; cherry, wild *(Prunus avium)*, elm, english (*Ulmus procera*); elm, wych (*Ulmus glabra*); hawthorn (*Crataegus monogyna)*; hazel (*Corylus avellana)*; holly (*Ilex aquifolium)*; lime, small-leaved (*Tilia cordata*); maple, field (*Acer campestre*), rowan (*Sorbus aucuparia)*; service tree, true (*Sorbus domestica*), oak, english (*Quercus robur*), oak, sessile (*Quercus petraea*), oak, pedunculate (*Quercus robur*), sycamore (*Acer pseudoplatanus*), whitebeam (*Sorbus aria*); willow, crack (*Salix fragilis*); willow, goat (*Salix caprea*); willow, white (*Salix alba*); Yew (*Taxus baccata*).

The native species of Wales differ in their optimal soil conditions for growth including soil pH, nutrient status and soil moisture (Table 11, Table 11). This highlights the importance of assessing conditions at the intended planting sites to ensure suitable species are planted. Optimal conditions (BSBI, 2020) are presented in Table 10. Preferred conditions for each species are presented as Ellenberg values averaged across the British and Irish distribution of each species and presented in Table 11.

It is important to consider site conditions and the aim of the woodland when choosing the most appropriate species to plant. If carbon sequestration is the most important factor and it is possible to leave the plantation for an indefinite period, then choosing species that generate the highest total biomass would be preferable. However, if the timber is to be harvested then choosing species that grow quickly (i.e. those that achieve the highest mean annual increment (MAI) in as few years as possible)

would be the most appropriate. In Table 10 maximum growth rates expressed as MAI are presented as a range encompassing all yield classes for each species. Maximum MAI can be expected to fall between the values presented in Table 10 on any site that is conducive to growth. Where MAI is unknown it might be possible to use data from similar species as suggested by (Kindermann *et al.*, 2013).

The most appropriate altitude range for species are displayed in Table 11. However this is only a range of recorded occurrence and does not inidicate at what altitude a species will thrive, oak woodlands for instance are rare above 300m in Britain (Jones, 1959). Furthermore, other factors associated with altitude could compromise vigourous growth and regeneration at high altitidues, for example wind-cut alder and hazel are seen on south-west slopes in North Wales.

The rate at which trees sequester atmospheric CO_2 will vary by species, for instance for stands in temperate humid regions aged 0-20 years oaks accumulated 9.5 CO_2 ha^{-1} yr^{-1} whilst mixed stands of other broadleaves accumulated 11.8 t CO_2 ha^{-1} yr^{-1} (Bernal *et al.*, 2018). Carbon fixation also depends upon the tree size with larger older trees fixing significantly larger amount of carbon compared to trees which are smaller in size (Stephenson *et al.*, 2014) which may be an incentive to protect existing, mature woodlands.

Table 10. Characteristics of broadleaf trees native to Wales.

Species	Optimum Soil pH	Age of maximum MAI: Year ($m^3\ ha^{-1}\ y^{-1}$) *	Soil moisture preferences	Nutrient requirements	References
Alder *Alnus glutinosa*	4.2 – 7.5	30 – 40 (4.4 – 14.6)	Water demanding, cannot control transpiration. Roots adapted to wet conditions.	Can grow on soils of varying nutrient status. Can fix atmospheric N_2. Nodulation most successful at pH 5.5-7.2.	(Claessens *et al.*, 2010)
Ash *Fraxinus excelsior*	5 – 7.5	40 – 45 (3.9 – 12)	Prefers freely drained soils.	High demand, especially nitrogen, calcium, magnesium and phosphorus. <30% base saturation precludes ash.	(Forestry Commission, 1971; Dobrowolska *et al.*, 2011)
Aspen *Populus tremula* L.	5 – 8.5	30 – 40 (3.9 – 14.0)	Tolerates high water content soil but prefers freely drained mineral soils.	Can grow on soils with poor nutrient status.	(Meikle, 1984; Worrell, 1995)
Beech *Fagus sylvatica*	3.5 – 8.5	80 – 95 (5.9 – 10.0)	Favours well drained soils and does not tolerate flooding. Cannot grow on wet clays.	Young beech can be highly sensitive to water and nutrient competition, especially with grasses. Growth response of beech seedlings to soil fertility varies in relation to radiation.	(Forestry Commission, 1971; Minotta & Pinzauti, 1996; Coll *et al.*, 2004; Houston Durrant *et al.*, 2016)
Birch *Betula pendula and Betula pubescens*	3 – 6[5]	40 – 45 (3.9 – 12)	Silver birch (*Betula pendula*) grows best on well-drained soils. Downy birch (*B.pubescens*) tolerates damper poorly drained conditions. Intolerant of drought. Seedlings were unable to access water potential of the soil below 10 cm.	Optimum proportions of mineral nutrients were reported at 100 N: 65 K: 13 P: 7 Ca: 8.5 Mg. Optimal concentration of N is 55-110 mg l^{-1} but has only a small effect when optimal proportions are present. *B.pubescens* extracted phosphate mainly from the first 10 cm of freely draining brown podzol. There are Zinc tolerant genotypes.	(Forestry Commission, 1971; Atkinson, 1992; Beck *et al.*, 2016)
Cherry *Prunus padus*	5 – 8	unknown	Very wet and very dry soils are unsuitable. It is typical of partially drained sites. Adults have drought tolerance.	Best growth is obtained in soil that is moderately nutrient rich.	(Leather, 1996; Uusitalo, 2004)
Elm *Ulmus procera* and *Ulmus glabra*	4.7 – 7.5	80 – 95 (5.9 – 10.0)	Has a preference for water rich soils, found near rivers, streams and floodplains. Tolerates waterlogging and drought stress.	Wych elm monocultures had higher SOC than *F.excelsior, F.sylvatica* and *Q.*robur and its leaf litter has been suggested to reduce nitrate leaching. Intolerant to low N availability. Best suited to fertile sites.	(Thomas *et al.*, 2018)
Hawthorn *Crataegus monogyna*	4.4 – 7.8	unknown	Prefers free draining sites	Can tolerate infertile and shallow soils.	(Good *et al.*, 1990; Pasiecznik, 2008)
Hazel *Corylus avellana*	4.31 – 6.19	unknown	Moist but well-drained.	Grows best on fertile nutrient rich soils.	(Adiloglu & Adiloglu, 2004; Enescu *et al.*, 2016)
Holly *Ilex aquifolium*	4.1 – 5.8	unknown	Generally considered intolerant of very wet. But has been found in some sites where, except for the top few centimetres, waterlogging is permanent. Seedling mortality occurs mainly through drought.	Can grow on acid heath podsols. Soils can be highly varied with Across sites key measurable factors such as Loss on ignition, base saturation and extractable phosphorous ranged from 2.9-76.6%, 12-100% and 0.3-3.9 mg/100g respectively at a soil depth of 15 cm.	(Peterken & Lloyd, 1967)
Lime, small leaved *Tilia cordata*	3.8 – 6.1	60 – 100 (5 – 11)	Can occur on well-drained soil and waterlogged sites but will perform poorer on the latter. Deep rooting and drought tolerant.	Where *T.cordata* is dominant, on deep grey-brown earths, soil nutrients are generally higher than on weakly podzolized soils where it will still grow with reduced vigour. Soluble phosphate ranges from 70-270 mg kg^{-1} on the former and 13-70 mg kg^{-1} on the	(Pigott, 1991; De Jaegere *et al.*, 2016)

				latter. Growth on young trees is most severely reduced on podzols and shallow rendzinas.	
Rowan *Sorbus aucuparia*	<5.5	unknown	Grows well on well-drained soils; absent from wetlands. Poor drought tolerance	Suggested to grow on nutrient poor soil but with reduced vigour.	(Raspé *et al.*, 2000)
Oak, English *Quercus robur*	4.9 – 5.4[15]	65 – 80 (65 – 80)	More tolerant to considerable waterlogging and even flooding. Preference for moist heavy soils. Drainage changes caused by blocked drains or severe drought frequently cause root-rot. *Q.robur* may be more sensitive to such changes than *Q.petraea*.	*Q.robur* prefers more basic soils rich in mineral nutrients whereas *Q.petraea* will prefer more acid soils. *Fagus* and *Quercus* often compete, the former is more responsive to nutrient concentration so on low nutrient soils *Quercus* has an advantage.	(Jones, 1959; Forestry Commission, 1971)
Oak, Sessile *Quercus petraea*	4.3 – 5.0[15]		Prefers well drained soils and is intolerant of waterlogging.	On the most nutrient-deficient soils; podsols or gley-podsols on siliceous rocks *Q.petraea* is usually the only species when soil is well drained.	(Jones, 1959)

* MAI is dependent on yield class of the stand which will be influenced by the site characteristics.

Table 11. Summary of the optimal conditions of broadleaves native to Wales from (BSBI, 2020).

Species	Ellenberg Values					Mean Jan Temp (°C)	Mean July Temp (°C)	Annual Precipitation (mm)	Altitude range (m)
	Light	Moisture	Reaction	Nitrogen	Salt Tolerance				
Alder *Alnus glutinosa*	5	8	6	6	0	3.5	14.5	1100	0 – 470
Ash *Fraxinus excelsior*	5	6	7	6	0	3.6	14.7	1069	0 – 585
Aspen *Populus tremula* L.	6	5	5	6	0	3.3	14.5	1102	0 – 640
Beech *Fagus sylvatica*	3	5	5	5	0	3.5	14.7	1061	0 – 650
Birch, silver *Betula pendula*	7	5	4	4	0	3.3	14.6	1073	Upper limit unknown
Birch, downy *Betula pubescens*	7	7	4	4	0	3.4	14.4	1123	0 – 685
Cherry *Prunus padus*	5	6	6	7	0	2.7	13.8	1240	0 – 650
Elm, English *Ulmus procera*	5	5	8	6	0	3.7	15.8	819	Lowland
Elm, Wych *Ulmus glabra*	4	5	7	6	0	3.4	14.7	1056	0 – 530
Hawthorn *Crataegus monogyna*	6	5	7	6	0	3.6	14.7	1073	0 – 610
Hazel *Corylus avellana*	4	5	6	6	0	3.5	14.6	1094	0 – 640
Holly *Ilex aquifolium*	5	5	5	5	0	3.6	14.7	1079	0 – 600

Lime, small leaved *Tilia cordata*	5	5	6	5	0	3.5	15.5	873	0 – 600
Rowan *Sorbus aucuparia*	6	6	3	4	0	3.4	14.4	1128	0 – 870
Oak, English *Quercus robur*	7	5	5	4	0	3.5	14.7	1049	0 – 450
Oak, Sessile *Quercus petraea*	6	6	3	4	0	3.5	14.6	1136	0 – 450

Environmental impact of tree planting upon organo-mineral soils

Once trees are established, tree canopy cover generally protects soils from erosion however there are significant risks of erosion during planting especially when existing vegetation has been removed and soil is disturbed but before the newly planted trees are established. Similarly, during management and harvesting of commercial forestry there is a potential risk of damage to soil structure and erosion. Soils are a consideration of the Environmental Impact Assessment (Forestry) (England and Wales) Regulations 1999 which may require a permit for afforestation and deforestation projects dependent upon the size and location of activity. The UK Forestry Standard also provides detailed guidelines and recommendations for the sustainable management of soils by the forestry industry (Appendix 2A) (Forestry Commission, 2017).

Soil properties

Impacts of forestry upon organo-mineral soil properties include:

- Soil erosion: erosion risk is affected by the physical, biological and chemical properties of soils; the relief of the land, the vegetation type and cover and the timing and use of management practices (Knox *et al*., 2015). In the uplands, shallow organo-mineral soils on steep slopes can be particularly vulnerable and erosion risk is highest on steep slopes when soil is bare or there is loss of vegetation ground cover and soils are compacted or weakly structured. Water-induced erosion is a particular risk for Welsh upland soils and is thought to affect a greater area of land than wind erosion (Owens *et al.*, 2006). During heavy rainfall events, when soil saturation is reached, soil surface structure may be degraded by the impact of rainfall (Owens *et al.*, 2006). This threat is likely to increase in the future given predicted increases in frequency and severity of extreme rainfall events with climate change (IPCC, 2019). Estimates of soil erosion in Wales are higher than for the rest of the UK, which probably reflects the greater proportion of soils on sloping land exposed to high rainfall (Evans *et al.*, 2015).
- Soil Compaction: Ground preparation and harvesting operations are the main causal factor of compaction with forestry and risk is greatest when soil is wet. Compaction impacts important physical and biological soil processes including water infiltration, root development and activity of soil biota. This can impair tree growth and increase surface runoff and loss of nutrients and sediment. For example, Vanguelova *et al.*, (2017) found that use of machinery for stump removal on peaty gley soil increased loss of carbon, nitrogen and base cations.
- Loss of SOM: Accumulation of SOM is favoured in low disturbance systems with organic matter inputs e.g. from leaf litter and roots. Organo-mineral soils are particularly vulnerable to loss of organic matter (National Soils Resources Institute, 2004). SOM is lost from organo-mineral soils by mineralisation and oxidation processes which can be caused by a range of management practices including drainage and cultivation.

Hydrological effects

Afforestation of organo-mineral soils may influence catchment hydrology as water is lost from trees by two main processes (i) transpiration water taken up by tree roots and lost through leaf stomata (ii) interception of rainfall by tree canopies and subsequent evaporation. Water loss from trees through these processes (collectively evapotranspiration) is influenced by many environmental factors including soils, temperature, wind and rainfall as well as forest characteristics such as species composition, age and structure.

Evapotranspiration rates differ between broadleaf and coniferous forest partly due to physiological differences between these forest types. Interception is generally higher from evergreen conifers due to their continual forest canopy throughout the year including during winter months when wet and

windy conditions are more frequent. Deciduous conifers such as larch (*Larix sp.*) may also have high interception due to their fine branch structure (Reynolds *et al.*, 1989). Based on UK studies it has been estimated that conifer stands may intercept between 25-45% of annual rainfall whilst broadleaves may intercept 10-25% (Calder *et al.*, 2003) and that these proportions generally remain consistent irrespective of total annual rainfall values (Nisbet, 2005). Previous studies investigating the impacts of afforestation on water yield in Welsh upland catchments have shown that due to the exceptionally wet and windy climate, values for rainfall interception by forests in these areas are amongst the highest in the UK (Marc & Robinson, 2007). For example (Calder, 1990) reported that up to 690 mm of rainwater may be lost by interception from mature Sitka spruce forest in upland Wales.

Whilst water loss by transpiration and evaporation is generally consistent amongst conifer species, broadleaf species show more variation. In particular, differences in canopy structure between broadleaves affect interception with more rainfall intercepted by species such as oak and beech which have dense canopies compared to those with less dense canopies such as ash and birch (Nisbet, 2005). There is also variation in transpiration rate of broadleaf species. Some broadleaves have exceptionally high transpiration rates such as poplar and willow which can support transpiration rates exceeding 500 mm annually when grown on wet soils (Hall *et al.*, 1996).

The effect of afforestation upon water yield is also partly determined by the vegetation type which is being replaced. Both upland grassland and heathland, which occupy the majority of the area of organo-mineral soils in Wales can sustain high total evapotranspiration rates (Table 12). Nether-the-less, studies of overall impacts of afforestation upon water yield have shown that replacement of grassland or moorland vegetation with coniferous forest generally results in reduced water yield of a catchment (Marc & Robinson, 2007; Forestry Commission, 2017). Mature coniferous forest with a closed canopy may cause the greatest reduction in water yield whilst newly planted, and felled forest may have similar water yield to moorland and grassland due to the smaller and less dense canopy of young trees (Forestry Commission, 2017).

Catchment scale effects of commercial conifer afforestation have been studied in detail using a long-term study at Plynlimon, mid Wales which was established in 1969. The Plynlimon catchment has three main soil types of which one is organo-mineral (peaty gley/peaty gley podzol) (Bell, 2005). The Plynlimon study compared the afforested catchment of the River Severn with the sheep grazed grassland/moorland catchment of the river Wye. The sources of both rivers are approximately 3 km apart with a 10 km^2 headwater area thus allowing similar environmental conditions within the upland catchments. Forest evaporative water loss recorded at Plynlimon is very high due to the wet and windy climate at this location which is typical of welsh upland areas (circa 220 rain days per year and annual rainfall up to 2500 mm). When monitoring began in 1969 the Severn catchment was 67% forested with commercial conifer plantation mainly comprised of Sitka spruce (*Picea sitchensis*), Lodgepole pine (*Pinus contorta*), and Norway spruce (*Picea abies*) with stands aged approximately 30 years. Measurements of the 30-40 year old forest (1972-1976) at which point the catchment had 70% forest cover, showed an annual water loss of 688 mm compared to 443 mm by the grassland Wye catchment (Marc & Robinson, 2007). The authors was estimated that with 100% forest cover the Severn catchment would lose 810 mm annually by evapotranspiration (182% of the loss from grassland). However, as the forest aged, water loss by evaoptranspiration was reduced as a result of changes to tree physiology. Estimates suggested that when the forest was 40-50 years old, the forested Severn catchment lost 667 mm via evaoptranspiration compared to 458 mm from than the adjacent grassland catchment (146% of the loss from grassland) (Marc & Robinson, 2007). Further forest harvesting in the Severn catchment reduced forest cover to 30% of the catchment area at which point water loss was similar between forested and grassland catchments (Marc & Robinson, 2007).

The Plynlimon catchment studies also demonstrated that afforestation of Welsh upland catchments may reduce stream flow by 10-15% compared to grassland. It has been estimated that potential water yield may be reduced by 1.5-2% for every 10% of mature coniferous forest within a catchment (Forestry Commission, 2017).

To date studies of afforestation upon catchment scale water yield have focused upon coniferous forests and there is no available data to show the impact of afforestation with broadleaf woodlands in upland areas. However, it is expected that in upland environments broadleaf afforestation would have less effect on water yield than coniferous forestry due to lower rates of water loss by evapotranspiration from broadleaves (Table 12) (Nisbet, 2005). Afforestation will also impact other aspects of catchment hydrology such a low and peak water flow, downstream flood risk and water quality. These effects are variable between sites and have been summarised in detail by the Forestry Commission, (2017).

Table 12. Typical range of annual evapotranspiration water losses (mm) for different land covers receiving 1000 mm annual rainfall (from Nisbet, 2005). In windy Welsh upland areas interception and transpiration values will be at the high end of the ranges shown.

Land cover	Transpiration (mm)	Interception (mm)	Total evaporation (mm)
Conifers	300-350	250-450	550-800
Broadleaves	300-390	100-250	400-640
Grass	400-600	-	400-600
Heather	200-420	160-190	360-610
Bracken	400-600	200	600-800
Arable*	370-430	-	370-430

*Assuming no irrigation

Ecosystem services

Woodlands on organo-mineral soils provide a range of ecosystem services which are summarised in Table 13. Bol *et al.*, (2011) compared ecosystem service provision between the main organo-mineral soil land use types in Wales which are grassland (rough, acid and bracken), forest (deciduous and coniferous), heath/montane habitat (dense and open) and cropland (arable and horticulture). This comparison suggested that woodland (in particular broadleaved woodland) is able to deliver the highest number of ecosystem services of all the organo-mineral soil land use types that were compared and that ecosystem service provision was generally supported to a higher degree than by other land cover types (Bol *et al.*, 2011).

The main negative impact of land use conversion to woodland was loss of food provisioning services. However, over 97% of the organo-mineral soil land area in Wales is classed as disadvantaged (with over 92% severely disadvantaged) which means the capacity of the land for food production is lower than the national average. This may be due to steep relief of land and high rainfall which may limit farming options (Welsh Government, 2019). Much of this low productivity land currently supports low density upland sheep farming (Welsh Government, 2019). For the 1.1% of the Welsh organo-mineral soil area which is currently under arable land use, afforestation would result in a larger loss in food production capability compared with conversion of upland grassland.

However, it is important to note that benefit to ecosystem services provided by afforestation are site specific and also depend upon the woodland type and suitability for the location, management and purpose. A recent report by the Natural Capital Committee emphasised the importance of assessing site suitability (for example location, current land use and environmental condition) before planting as afforestation of unsuitable sites may have adverse effects on ecosystem service provision including soil, water and air quality, recreation and biodiversity (NCC, 2020).

Table 13. Provisioning, regulating, supporting and cultural ecosystem services which may be provided by deciduous and coniferous forest on organo-mineral soils. The influence of forestry on each ecosystem service provision may be positive (blue text) or negative (red text). Size of the effect is categorised as none, low, medium or high. Adapted from Bol *et al.*, (2011). This is a non-exhaustive list of the ecosystem services which may be provided.

Ecosystem Service	Deciduous forest	Coniferous forest
Provisioning services:		
Food	None	None
Fibre	High	High
Fuel	High	High
Genetic resources	Medium	Low
Refugia	High	Medium
Water provision	Low	Low
Regulating services:		
Global climate	Medium	Medium
Regional climate	Medium	Medium
Local climate	Medium	Medium
Flood Hazard regulation	Medium	Medium
Erosion control	Medium	Medium
Pollination services	High	Medium
Noise regulation	Low	Low
Soil quality	High	Medium
Air quality	High	Medium
Water quality	High	Medium
Supporting services:		
Primary production	High	High
Nutrient cycling	High	High
Soil formation	High	Medium
Water cycling	Medium	Medium
Cultural services:		
Recreation	High	High
Tourism	High	High
Cultural heritage	High	Medium
Aesthetics	High	Medium

Biodiversity

In the UK, a large proportion of woodland species have suffered population decline, particularly those which are reliant upon structurally diverse native woodland (Hayhow *et al.*, 2016). The recent State of Nature report for the UK showed that over half of the 1,285 species which depend upon woodland have experienced population decline and one in 10 woodland species are threatened by extinction (Hayhow *et al.*, 2016). In Wales, many native broadleaf woods were felled and replanted with conifers in the past century (Warren-Thomas & Henderson, 2017). Planting, protecting and restoring native woodlands will be beneficial for supporting native biodiversity (Welsh Government, 2018).

Whilst woodlands can be beneficial for supporting biodiversity, effects of afforestation upon biodiversity are site specific and it is clear than in some sites, afforestation could negatively impact the biodiversity of the site or adjacent sites (Wilson *et al.*, 2014; Burton *et al.*, 2018). Over 23% of the area of Welsh organo-mineral soils are currently designated as SSSI sites which may be due to specialist plant or animal communities. Where afforestation may affect a SSSI site, the ecological impact must be fully

assessed to prevent detrimental impacts upon SSSI habitats or species. Similarly, changes to land use which will impact upon the environmental quality of SAC, SPA or NNR land should be avoided. Considering the impact of afforestation upon biodiversity of non-designated habitats is required through Environmental Impact Assessments (Forestry) (England and Wales) Regulations 1999.

Suitability of organo-mineral soil sites for afforestation may also be restricted by the requirement to avoid afforestation where this may impact deep peat habitats (Forestry Commission, 2017; UKWAS, 2018). In upland areas of Wales, organo-mineral soils often occur within a mosaic of deep peat or adjacent to deep peat sites (Map 2). These adjacent areas of deep and shallow peat are hydrologically linked therefore afforestation of organo-mineral soils in these locations could lead to drying of the adjacent deep peat habitats which in turn could impact deep peat vegetation communities, and functions such as carbon storage. Afforestation of areas adjacent to deep peat may also lead to the direct impacts on deep peat habitats for example the reseeding of trees into drying peat bogs (Campbell & Robson, 2019).

Conclusions

- Organo-mineral soils are a nationally important carbon stock. With the exception of deep peats they hold more carbon per unit area than any other UK soils. Organo-mineral soils are estimated to hold 19% of the soil carbon stock to depth in Wales (Scottish Executive, 2007). Organo-mineral soils are also particularly vulnerable to erosion because they are often located on steep upland slopes.
- Management practices which physically disrupt soil structure and remove vegetation cover increase the risk of degradation of organo-mineral soils. Physical soil disturbance increases the rate of soil organic matter mineralisation and decomposition leading to increased rates of carbon loss to the atmosphere as CO_2 or to water sources as dissolved organic carbon or particulate organic carbon. Organo-mineral soils may lose carbon at a much faster rate than carbon can be sequestered, therefore preventing soil carbon loss is a priority.
- There is uncertainty regarding future climate change for Wales and how this will impact organo-mineral soils however it is likely that climate change will increase erosion risk. Increased severity and frequency of extreme weather events such summer drought, intense winter rainfall and storm events are likely to increase the risk of erosion particularly on exposed upland areas with steep slopes. Soil disturbance, exposure of bare soil and reduced vegetation cover during ground preparation for forestry and at early stages of planting could further increase the risk of soil erosion at these sites.
- There is still uncertainty regarding the impact of afforestation on the carbon stocks contained in organo-mineral soils. It is clear that activities associated with afforestation such as planting, management and harvesting, disturb soil and may cause soil carbon loss during first rotation forestry.
- Recent research by the forestry industry has suggested that when moderate and high productivity trees are planted using low soil disturbance methods it is possible that afforestation may result in little or no net change in soil carbon when the full cycle of forest growth is considered over two forest rotations. However, this is based on limited evidence and there no data to show the longer (> 100 year) effects of afforestation upon organo-mineral soil carbon stocks. The extent to which these carbon losses may be balanced by carbon accumulation in the litter layer and forest floor layers over multiple forest rotations and contribute to recovery of soil carbon stocks requires further attention.
- Forests are able to store more carbon within plant biomass than other vegetation types due to the larger amount of plant biomass that can be produced by trees per unit area of land. Therefore conversion of organo-mineral soil land use for example from moorland or grassland to forestry would potentially increase the vegetation carbon store. However, to determine contribution of the vegetation carbon store to the total net effect of forestry upon carbon sequestration the long-term

fate of wood products from forestry must be considered. Use of harvested wood for short term products for example fuel results in rapid release of stored carbon back into the atmosphere and thus provides no benefit for carbon sequestration. In comparison, wood products used for long term purposes such as construction will provide a longer-term carbon store.

- To date, studies of afforestation upon organo-mineral soils have focused upon commercial forestry with coniferous species, predominantly Sitka spruce. Long term studies are essential for understanding soil carbon change over time however guidelines for forest management have changed significantly in the last 40 years with more focus on environmental protection such as erosion mitigation. Therefore, many of the long-term studies are partly based on outdated management practices and are less relevant for understanding impacts of modern forestry upon soils. No studies have examined the long-term impacts of afforestation upon organo-mineral soils for longer than 100 years however these studies are essential for fully understanding impacts of afforestation upon soil carbon.
- The impact of organo-mineral soil afforestation with broadleaf species upon soil carbon has not been quantified and requires further study. Additionally, comparison of the impact of broadleaf forest management practices including differences in planting methods upon organo-mineral soils is necessary to inform future best practice.
- Afforestation reduces water yields from catchments due to increased canopy interception of rainfall and evapotranspiration compared with grassland. Therefore it is important to consider the effects of upland afforestation upon water supply particularly given the predicted reductions in summer rainfall with climate change.
- Woodlands can support a range of ecosystem services however effects of land use change upon ecosystems services are site specific and should be carefully considered on a site by site basis. In a suitable location, native broadleaf woodlands are of high value for biodiversity provision.
- Over 23% of the land area of welsh organo-mineral soils is currently classified as SSSI land. Any change to land management on or near these sites must be carefully assessed on a site by site basis. The same applies to land use of organo-mineral soils in or near to SAC, SPA and NNR areas.
- In many areas of the Welsh uplands organo-mineral soils are in close proximity to deep peat soils (>50 cm deep). Due to hydrological connection between these soils, afforestation of organo-mineral soils in these locations is likely to impact the hydrology of the deep peat. At these sites afforestation could also adversely affect deep peat vegetation communities and associated biodiversity.

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Appendix

Table 1A. Definitions of Soil wetness classes from (Hodgson, 1997). The number of days specified is not necessarily a continuous period.

Wetness class	Definition
I	The soil profile is not wet within 70 cm depth for more than 30 days in most years*.
II	The soil profile is wet within 70 cm depth for 31-90 days in most years or, if there is no slowly permeable layer within 80 cm depth, it is wet within 70 cm for more than 90 days, but not wet within 40 cm depth for more than 30 days in most years.
III	The soil profile is wet within 70 cm depth for 91-180 days in most years or, if there is no slowly permeable layer within 80 cm depth, it is wet within 70 cm for more than 180 days, but only wet within 40 cm depth for between 31 and 90 days in most years.
V	The soil profile is wet within 40 cm depth for 211-335 days in most years.
VI	The soil profile is wet within 40 cm depth for more than 335 days in most years.

*'In most years' is defined as more than 10 out of 20 years.

Table 2A. UK forestry standard guidelines for sustainable soil management

Guideline No.	Sustainable management
20	Ensure the removal of forest products from the site, including non-timber products, does not deplete site fertility or soil carbon over the long term and maintains the site potential.
11	Maintain adequate brash mats throughout extraction operations.
10	On sites vulnerable to compaction and erosion, consider the weather and aim to carry out operations during dry periods; plan ahead for changes in the weather that could affect site conditions.
26	Avoid burning brash and harvesting residues unless it can be demonstrated that it is a management necessity, all the impacts have been considered, and the necessary approvals obtained.
15	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.
9	Minimise compaction, rutting and erosion during forest operations by selecting the most appropriate working method for site conditions; monitor operations and modify, postpone or stop procedures if degradation starts to occur.
13	Minimise the soil disturbance necessary to secure management objectives, particularly on organic soils.
24	Avoid establishing new forests on soils with peat exceeding 50 cm in depth and on sites that would compromise the hydrology of adjacent bog or wetland habitats. Note: Woodland creation on certain sites where deep peat soils have historically been highly modified may be considered, provided that it complies with the relevant country policy.
14	Consider the potential impacts of soil disturbance when planning operations involving cultivation, harvesting, drainage and road construction.
20	Ensure the removal of forest products from the site, including non-timber products, does not deplete site fertility or soil carbon over the long term and maintains the site potential.
25	Consider the balance of benefits for carbon and other ecosystem services before making the decision to restock on soils with peat exceeding 50 cm in depth.
18	On steep slopes where there is a risk of slope failure or serious erosion, consider alternatives to clearfelling.
5	Minimise the use of pesticides and fertilisers in accordance with Forestry Commission and Forest Service guidance.