Forest Research

Understanding the GHG implications of forestry on peat soils in Scotland

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Report compiled by FR staff for FC Scotland, October 2010.

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Executive Summary

1. Peat soils in Britain have accumulated large stocks of organic carbon which are altered by changes in land use, including forestry, that disturb the soil or the water table. Drainage or cultivation usually results in loss of soil organic carbon (SOC) by carbon dioxide (CO_2) efflux, by particulate erosion or dissolved in rainwater drainage and runoff. Conversely, forest growth can lead to an accumulation of SOC, through litter formation and incorporation of organic matter into the soil, as well as the C stocks accumulating in the trees.
2. FC Scotland issued draft interim policy guidance on woodland creation on deep peats in January 2010 of a general presumption against new woodland creation on soils with peat exceeding 50 cm in depth. This report was commissioned to help develop a better understanding of the greenhouse gas (GHG) implications of forestry on peat soils in Scotland.
3. Understanding the consequences of forestry activity on peat SOC and GHG balance depends upon four important aspects: the type of peat soil and proportions of different SOC fractions, whether previously planted and how prepared or cultivated, the level of disturbance during planting, and the modification to the water table depth. Understanding the overall GHG balance of forests on peat also depends upon the rate of CO_2 taken up by the trees during growth, and the accumulation or use of harvested wood products (HWP) and their possible net emissions benefits through substitution for fossil fuel intensive materials and energy.
4. Measurements in Scottish forest soils have shown that in the top 1 m the average SOC of shallow peat soils (e.g. peaty gleys and podzols, <40 cm peat layer) is 350 t C ha^{-1}, and that in deep peat soils (>40 cm peat layer) is 510 t C ha^{-1}.
5. Approximately 1,500 km^2 or 11% of present Scottish forests are on deep peat soils, and 5,900 km^2 (44%) on shallow peat soils. Deep peats are less widely distributed in Scotland compared to shallower peat soils and are concentrated in Dumfries and Galloway, the Grampian Highlands and NE areas of mainland Scotland.
6. In Scotland the total C stock in deep peat soils (blanket and basin peat types) under forest is estimated to be 76 Mt C, 27% of the total SOC of forested shallow and deep peat soils (down to 1 m). While the amount of SOC below 1 m is not known, averaging across all deep blanket peat soils in Scotland suggests a maximum of an additional 35 Mt C. Restricting planting of new woodland in Scotland so that soils with >50 cm peat depth were not used would affect between approximately 7,400 and 10,300 km^2 although this includes areas with >1 m depth, already excluded from planting. The area of un-forested shallow peat soils is 3-4 fold larger, with substantially lower and less vulnerable soil C stocks per unit area.
7. It is very difficult to quantify the extent to which tree planting on peat soils results in loss of SOC and changes to GHG balances as there are very few measurements of simultaneous CO_2, methane (CH_4) and nitrous oxide (N_2O) fluxes from pristine or afforested peat soils either outside the UK or within the UK. The available data suggest that CO_2 efflux from the soil dominates in the total contribution to the GHG balance, although CH_4 emissions can be substantial in the wettest sites.
8. During planting or restocking some form of soil cultivation is usually necessary to ensure good tree survival and growth, for all apart from the most fertile of sites, with consequent losses of SOC. However, there are few data to help understand the impacts of soil disturbance on soil carbon stocks, soil water and aeration conditions and thus GHG fluxes. In particular, there is a lack of quantitative understanding of the effect of disturbance on SOC stability. Ground preparation practices have changed since the major forest expansion of 1960-80's, and older sites do not necessarily provide SOC and GHG balance data that are relevant to current practice. There is

therefore considerable uncertainty in the conclusions presented here, although there is sufficient overall understanding to inform policy recommendations.

9. Understanding the net C and GHG balance of a forest stand and the soil requires use of detailed process models, and these need to be sufficiently comprehensive to include CH_4 and N_2O fluxes, the effects of alterations to water tables and other disturbances, and be appropriate for organic soils in UK conditions. Appropriate soil GHG process models are only now becoming available, although their treatments of disturbance effects are presently limited. In addition, the overall GHG balance benefits of CO_2 uptake by the forest and production of timber and/or woodfuel should be considered. This will require linking existing robust forest C accounting models with appropriate soil models.
10. It is very probable that moderate and high productivity forests planted on shallower peat soils with limited disturbance provide a net C uptake over the forest cycle, because uptake of CO_2 by the forest exceeds emissions from soil decomposition. In addition, there is likely to be a substantial reduction in CH_4 emissions with afforestation, further improving the forest GHG balance, and the yield of HWP will also contribute substantial GHG emissions abatement benefits. On deep peats where tree growth is likely to be poor without substantial site modification, new drainage, cultivation or fertilization is usually required to achieve good growth from commercial conifer species. This would result in increased GHG emissions and SOC losses so that it is probable that the net GHG balance would be negative.
11. Therefore, in view of the large amount of soil C presently stored in deep peats in Scotland (relative to the land area involved), restricting new planting to shallower peats (<50 cm deep) with less potential C loss, and usually better tree growth conditions is a sensible precaution.
12. The net GHG balance benefits of restocking previously planted and disturbed deep peat sites are very different from those of new planting on peat soils which currently support non-woodland vegetation. If tree growth is likely to be good, and little further disturbance necessary, then SOC stocks may recover the losses during the first rotation, and the CO_2 uptake by the forest add additional GHG benefits. On wet and low fertility deep peat sites, where tree growth will be poor, restocking is likely to result in a continuing negative GHG balance. Such sites should be prioritized for open habitat restoration so that the continued loss of SOC will eventually be stopped.

1. Introduction

The forest cover in Scotland has increased substantially in the past 90 years from 5.6% of land cover to 17.2% (435,000 ha to 1.34 Mha; Forestry Commission, 2008). Much of the increase has been in upland areas, and therefore much of the forest has been established on peaty gley soils and other soils with high organic matter (OM) contents from peat deposits. There is increasing demand for improved management of peaty soils (including mires, peatbogs, fens and other peatlands), driven, in part, by interest in protecting stored ecosystem carbon (e.g. Byrne et al., 2004; Worrall et al., 2010). Disturbing peat soils for tree planting can lead to enhanced losses of organic carbon (see e.g. Jarvis et al., 2009). Given the importance of both maintaining existing soil C stocks and enhancing the capture of atmospheric CO_2 by forests and their soils as components of climate change mitigation strategies (e.g. Nabuurs et al., 2007; Reed et al., 2009), it is timely to assess the implications of tree planting on peat-containing soils. However, such an assessment needs to consider other routes for C loss than CO_2 efflux alone (e.g. Worrall et al., 2010) and emissions of other key greenhouse gases (GHG) because peat soils can be substantial sources of methane (CH_4) and in some cases of nitrous oxide (N_2O, e.g. Byrne et al., 2004). Importantly, the assessment also needs to consider the GHG balance benefits of utilising forest products outside the forest through the substitution effects of woody biomass reducing fossil fuel combustion and timber replacing fossil-fuel intensive man-made materials (see Matthews & Broadmeadow, 2009). In addition, forest operations frequently use fossil fuels although this is a minor component of the forest GHG balance in most situations examined (see Mason, Nicoll & Perks 2009). This set of linked and complex issues means that the recommendation in the draft 2009 UKFS Climate Change Guidelines "to consider the overall GHG balance" when making forestry decisions is a significant challenge. One response to concerns over safeguarding soil C in peat stocks is to reduce the peat depth limit for soils that should be considered for tree planting, and FC Scotland has issued interim policy guidance to that effect (see Appendix A), pending examination of the information available.

This report was commissioned by the FC Scotland in order to examine the issues above, with the specific aim "to help inform policy guidance through developing a better understanding of the greenhouse gas implications of forestry on peat soils in Scotland". The report was asked to consider 'peat soils' as those with peat layers of 5 cm thickness or more (see below for current FC definitions). The objectives are:

1. Confirm the current evidence base for the carbon content of peat soils.
2. Assess current knowledge of the expected greenhouse gas emissions from 'open ground' peat soils in Scotland's changing climate.
3. Assess current knowledge on the greenhouse gas implications of cultivation and drainage for initial tree establishment (new woodland creation) on peat soils.
4. Assess current knowledge on the greenhouse gas implications of cultivation and drainage for restocking existing woodland on peat soils.
5. Assess current knowledge of the full life cycle, <u>net</u> greenhouse gas emissions impacts of the main woodland types on peat soils.
6. In the light of available knowledge, comment on FCS's interim policy guidance on new planting and restocking on peat soils.
7. Identify gaps in knowledge and consequent research needs (with time-scales for outputs) for improvements to the evidence base.

This report is intended to be a concise summary of the information available for Scotland, with relevance to forestry. More general information about peat soils, their distribution and characteristics can be found in several recent reviews and reports, particularly the two ECOSSE reports (Smith et al., 2007; 2009); that for the RSPB (Lindsey, 2010) and Natural England (2010), although the latter only covers English peatlands. This report contains 3 main chapters. Chapter 2 (main author: Elena Vanguelova) summarises available information on the carbon[1] contained in peat soils in Scotland and their extent, and examining the uncertainties in that information. Chapter 3 (main authors: Sirwan Yamulki & Mike Perks) summarises the information on GHG fluxes from peat soils associated with forestry. Chapter 4 (main author: Tim Randle) presents some model calculations on the carbon balance of example forest stands on peat soils in order to compare GHG benefits of tree growth with possible losses due to peat disturbance. Chapter 5 summarises the key points, and the implications for the FCS interim policy guidance and Chapter 6 lists the main areas requiring more research.

Peat soil definition

The Forestry Commission soil classification (Kennedy, 2002, see also Appendix C) defines soil *groups* subdivided into *types* and indicated with a code. Particular soils may have additional *phases* indicated with an additional letter in the code. The classification makes a division between *shallow peaty soils* (organic matter depth < 45 cm) and *deep peats* (organic matter depth >45 cm). Shallow peaty soils are in groups 3 (Podzols), 4 (Ironpans), 5 (Groundwater gleys) and 6 (Peaty gley soils), and deep peats are in groups 8-14 (Appendix C). The classification defines that a soil may have a *peaty* soil phase (adding the letter ‘p’ to the type code): “a surface horizon containing more than 25% organic matter”. For soil types 3 and 5 to be described as 3p and 5p requires 5-45 cm of peat; for type 6p and 6zp the soil requires 25-45 cm of peat as these soils types already have >5 cm peat. For Ironpan soil types Kennedy (2002) suggests that 15-45 cm peat should be present for the p phase label to be assigned. Thus the ‘peat soil’ definition requested for this report of >5 cm depth is compatible with the FC soil classification for peats and peaty soils. However, the FC Scotland Interim Guidance limit of >50 cm does not match the FC soil classification threshold of >45 cm for deep peats (Kennedy, 2002). **Consideration should be given to using the FC deep peat threshold of >45 cm for the Guidance,** although it should be noted that Scottish soil maps (Soil Survey of Scotland) use a definition of >50 cm for deep peat soils and the World Reference Base for Soil Resources (WRB, 2006) classification of deep peats (histosols) has a threshold ≥40 cm.

2. Carbon in peat soils

2.1 Introduction

The estimation of the organic carbon contained in peat soils is difficult primarily because the depth varies very widely, and many surveys only assess soil characteristics in the range 30-80 cm. Clearly, if the soil has a deep peat layer extending below the survey limit, then the soil organic carbon content (SOC) will be substantially underestimated. Soils with deep layers of peat can extend to several metres (Smith et al., 2007; 2009), although these depths occur in peat bogs, which are not relevant for consideration as afforestation targets. A second serious problem is the accurate determination of the bulk density rather than estimates, and how bulk density varies with depth and spatially.

[1] As is usual, carbon (C) stocks in soils, trees etc., are expressed as mass of C, per unit area (e.g. t C ha^{-1}) or in total for a defined area (e.g. national stocks in Mt C). Gaseous and dissolved fluxes are referred to as mass of CO_2 per unit area per unit time (e.g. t CO_2 ha^{-1} y^{-1}); C stocks can be converted to stocks of CO_2 by multiplying by the ratio of the molecular weights, 44/12 (approx. x 3.67).

Recent estimates of SOC content in peat soils under forestry are given below, and C stocks of shallow and deep peats are compared and up-scaled to provide C stocks for Scotland under forestry. Uncertainties in up-scaling, due to the precision of soil mapping, are illustrated with two GIS case studies in Scotland. The fate of C stored in peat and mineral soils, its relation with nitrogen and the fluxes of dissolved organic carbon are also discussed in relation to SOC sensitivity and its potential loss due to disturbances.

2.2 Peat soil carbon content in Scotland

The recent BioSoil survey of 167 forested plots across GB can be used to assess C content in soils with different depths of peat (Vanguelova et al., in prep.), and in particular to assess the effect of varying the depth considered. The BioSoil survey assessed soil profiles down to 80 cm in 5 depths, and calculated C stocks from measurements of soil C% and bulk density. There were 36 *shallow peat* (peat depth <40 cm, as classified by WRB, 2006) and 14 *deep peat* (peat layer of ≥40 cm) plots in Scotland. The measurements from the five soil depths were used to extrapolate to 100 cm depth. **Total organic carbon stock down to 100 cm soil depth of *shallow peat soils* (e.g. peaty gleys and podzols) was 350 ±40 s.e. t C ha^{-1}, and the stock in *deep peat soils* was 510 ± 55 s.e. t C ha^{-1}** (Figure 2.1).

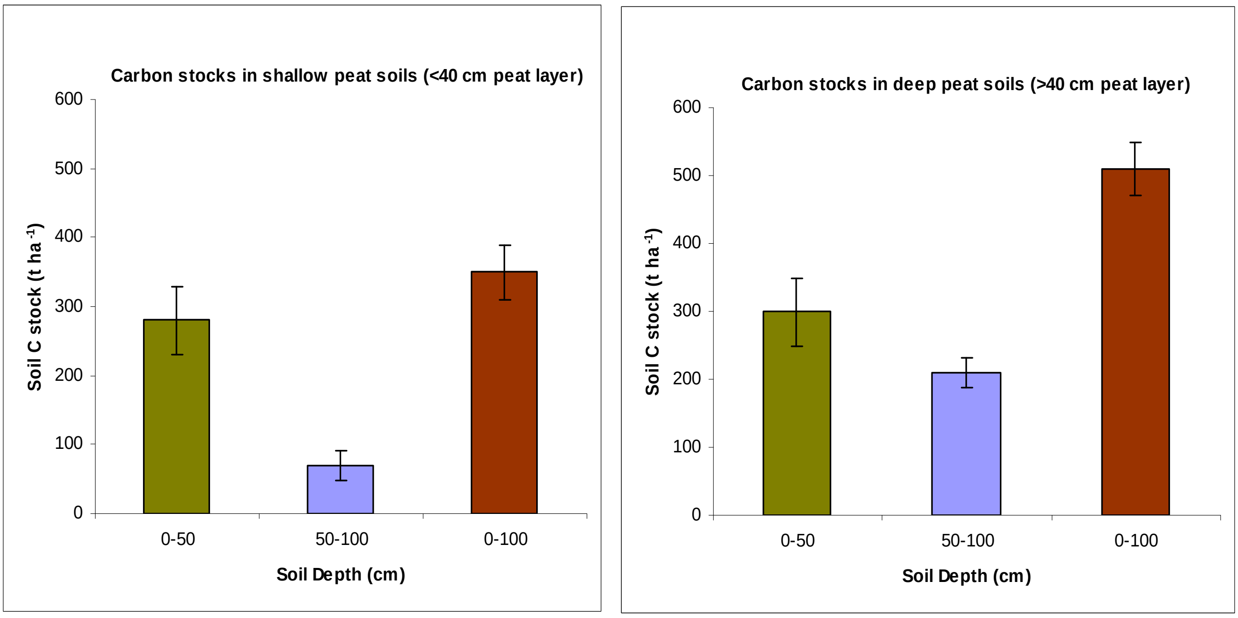


Figure 2.1. Soil organic C stocks in shallow (left) and deep (right) peat soils, in 0-50, 50-100 and 0-100 cm soil depth, data for 0-80 cm from BioSoil plots and extrapolations were carried out for soil C stocks between 80 and 100 cm.

Peat SOC content variation with depth was calculated from the BioSoil database (Figure 2.2). A simple extrapolation between measured soil depths suggests that **the amount of SOC between 0 and 50 cm is 300 t C ha^{-1} and between 50 and 100 cm is approximately 210 t C ha^{-1}**. This suggests that SOC stock between 50 and 100 cm depth in deep peats is much greater than SOC stock in mineral soils from 0-100 cm (total C stock ranging from 150 to 180 t C ha^{-1}).

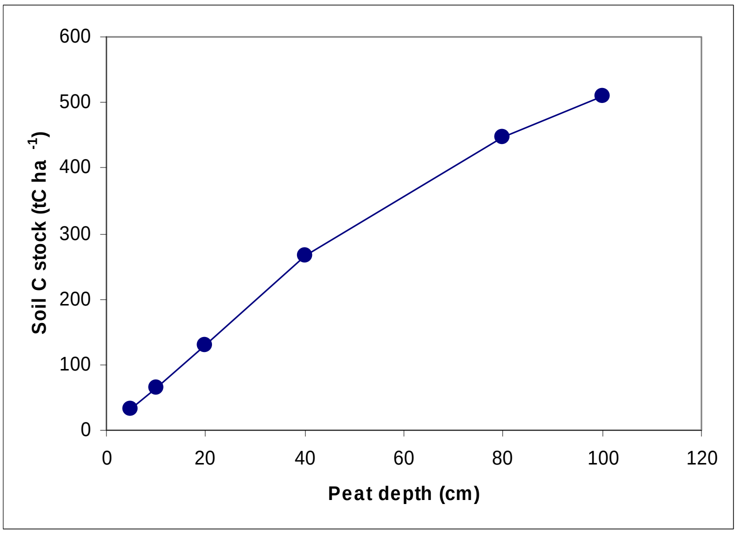


Figure 2.2. Average total peat C stocks related to peat layer depth (from BioSoil plots).

2.3 Forestry areas in Scotland on peat soils

Using data from the Scottish National Soil Map Classification, shallow peat soils in Scotland cover 38% of the total land area, while deep peats occupy 11.5% (Table 2.1). It should be noted that this classification uses a threshold of >50 cm for peat soils and the 'deep peats' were assumed to be only the blanket and basin peat types. The ECOSSE analysis (Smith et al., 2007) estimated exactly the same area for these deep peat types, although Patterson and Anderson (2000) reported a larger area of 10,562 km^2. Combining the above map classification with the Forestry Commission NIWT1 data (2001) suggests that coniferous forests planted on shallow and deep peats occupy 48% and 13%, respectively, of the total coniferous forest area in Scotland (Table 2.1). Broadleaved forests planted on shallow and deep peats occupy only 18% and 2%, respectively, of the total broadleaved area (Table 2.1). **In total approximately 1,500 km^2 or 11% of Scottish forests are on deep peat soils, and 5,900 km^2 (44%) are on shallow peat soils.** Approximately 17% of deep peat areas are forested.

Table 2.1. Total land area and land under Coniferous and Broadleaved woodland for each main soil type in Scotland (Scottish National Soil Map Classification (1:250K) Forestry Commission, NIWT1 2001 datasets used).

	Land area (km^2)		
Soil Type	Total area	Coniferous	Broadleaves
Brown Earths	13,385	1,425	740
Podzols and Ironpans	8,495	1,668	496
Surface-water gleys	10,096	1,090	309
Ground-water gleys	41	0	0
Peaty gleys and podzols	**30,094**	**5,540**	**369**
Deep peats*	**8,818**	**1,452**	**40**
Rankers and rendzinas	4,989	155	84
Other	2,861		
Total	*78,779*	*11,332*	*2,038*

*Deep peats means >50 cm peat depth, mostly blanket peat with basin peat types.

2.4 Peat C amounts under forestry in Scotland

The BioSoil data for C stock in shallow and deep peats was combined with the areas of these soil types under forests from Table 2.1 to estimate total peat soil SOC contents (down to 1 m depth). Under coniferous forests in Scotland there are 194 Mt C and 74 Mt C under shallow and deep peats, respectively, while for broadleaved forests the estimates are 13 Mt C and 2 Mt C, respectively. **The total C estimated as stored in shallow and deep peats under both forest types in Scotland (down to 1 m) is 207 and 76 Mt C, respectively.** However, deep peats obviously can have substantial soil C held below 1 m. The recent analysis of C stocks and areas of Scottish deep peat soils (Table 2.2, Chapman et al., 2010) which included estimates of SOC below 1 m depth suggested that there is an additional 48% of soil C below 1 m in blanket peats. Assuming this value is appropriate for forested deep peats produces **an estimated total SOC for deep peats under forest of 112 Mt C,** although this is probably an upper estimate because only a small proportion of total forest cover in Scotland will be on peats >1 m deep. Thus **if all areas defined as deep peats and currently under forest in Scotland were not replanted at the end of their current rotation, this would affect 11% of the currently forested area** (Table 2.1**), and an area which comprises between 27 and 35% of the soil C stock on all forested peat soils**.

Given current aspirations for an increase in woodland area, the question arises as to how much available area would be affected by restricting new planting to soils with <50 cm depth of peat. **The present area of blanket and basin deep peat *not* under forest (83%) is approximately 7,300 km^2** (Table 2.1) and part of this area includes peats >1 m depth, so under existing guidelines it would be excluded from forest planting (e.g. Patterson & Anderson, 2000). Compared with the much larger area of remaining unplanted and more productive shallower peaty gley and podsol soils (24,000 km^2), the removal of this potential deep peat area seems unlikely to have a large effect. Obviously, a proper analysis of land availability by soil type would need to take into account other existing land uses and multiple constraints, which is outside the present scope.

It is also possible to estimate the amount of SOC potentially protected by a restriction of not planting peat soils >50 cm deep. Chapman et al. (2009b) recently published estimates of C stocks of blanket and basin peats in Scotland totalling 781 Mt C (stocks <1 m depth, Table 2.2). Subtracting the soil C stock of 76 Mt C estimated above for these peats already under forestry, leaves 705 Mt C potentially protected by the restriction. Semi-confined peats are excluded from these calculations as they are not likely to be planted currently and not suitable for future planting.

Table 2.2 Estimated peatland areas and C stocks in Scotland for deep peat soils (>50 cm deep, from Chapman et al. 2009b).

Soil type	Area	C stock (< 1 m depth)	C stock (>1 m depth)	Total C stock
	(km^2)	(Mt C)		
Blanket peat	11,110	737	355	1091
Basin peat	730	44	77	120
Semi-confined peat	5,420	323	85	408
Total peat	*17,270*	*1104*	*516*	*1620*

However, it should be noted that the total area of blanket and basin peat types in Table 2.2 is some 34% higher than the comparable deep peat area in Table 2.1 because Chapman et al. (2009b) used more detailed soil mapping units than those available to Forest Research. It is clear that harmonisation of soil mapping at the same level of detail by different research institutions is necessary in order to reduce the uncertainties in area estimates for particular soil types, and hence in up-scaled soil C stocks in Scotland.

Spatial coverage of current shallow (peat layer < 50 cm) and deep peats (peat layer >50 cm) in Scotland in addition to forest cover on these soils is shown in the maps below (Figure 2.3a and b). Mapping is based on the area estimates listed in Table 2.1 above.

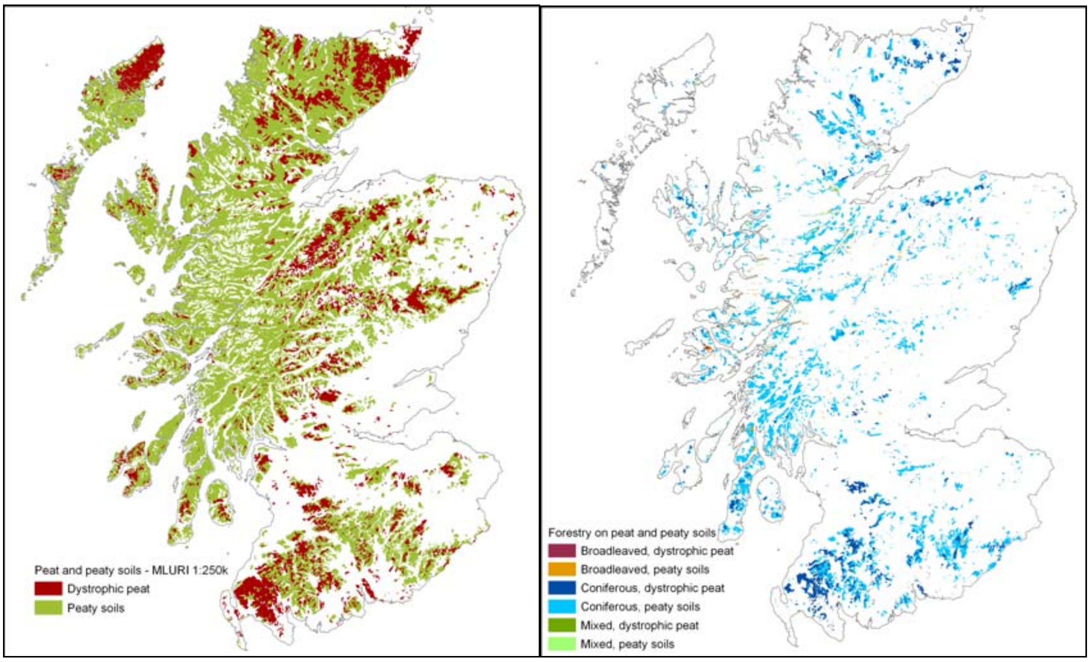


Figure 2.3 a and b areas of deep ('dystrophic') and shallow ('peaty') peat soils, and forest types in Scotland. (Data sources: Scottish National Soil Map Classification (1:250K) Forestry Commission, NIWT1 2001 datasets used).

It is evident in the map in Figure 2.3a that **deep peats, (those affected by the interim guidance), are less widely distributed in Scotland compared to shallower peat soils and are concentrated in Dumfries and Galloway, Grampian Highlands and NE areas of mainland Scotland**.

2.5 Uncertainty in peat soil C content from surveys

Estimates of the total organic C stock in soils for a region or country is calculated as:

Total C stock = area x depth x bulk density x soil C%

However, uncertainties exist in each of them. Relative uncertainties estimated from the Scottish soil survey for all different parameters used for soil C stock calculations are shown in Table 2.3 (ECOSSE report, 2009). Despite the very high number of observations of peat depth the percentage error in estimates is still high. The relatively small number of samples where bulk density is estimated also result in a large error of 8.3% despite the lower variability associated with bulk density measurements per horizon/depth suggested in the ECOSSE report, (Smith et al., 2009). In contrast to the Scottish soil survey, the estimation and up-scaling of the forest soil C stocks from BioSoil plots is subject to minimal uncertainties from depth, % C and bulk density as all three parameters were measured for each assessment plot. The main error in estimated forest soil C stocks for peaty and peat soils will be from the error associated with the area of these soil types in Scotland and the details of soil mapping.

Table 2.3. Basis of parameter estimates used in the calculation of total C stock in organic soils and their relative errors (ECOSSE report, Smith et al., 2009).

Parameter	**No. samples**	**Location**	**% Error in estimates**
Depth	~6000	Country-wide but some areas under-represented	7.2
% C	240	Country-wide but mainly surface (0 – 100 cm)	3.4
Bulk density	104	Country-wide but weighted towards NE Scotland and few deep samples (>200 cm)	8.3
Area	1455 polygons	Country-wide	4.5

A GIS study was undertaken to identify what is the relevant error by which soil C stocks differ due to different mapped proportions of soil type by National Scottish soil survey database compared with higher resolution FC soil mapping. Two contrasting areas were chosen, one with a higher proportion of peaty gley soils (Meallmore - mineral:peaty:deep peat soils = 2:94:5; total area = 1009 ha) and the other one with higher proportion of deep peats (Craik - 20:52:28; total area = 3881 ha). The comparison between the two soil maps for those two locations is shown below (Figure 2.4 a,b).

The areas for different soil types were calculated and grouped into three main soil types – mineral soils, peaty soils and deep peat soils. While the error (underestimate) by the lower-resolution map for the mineral soils is large, this is of less importance to C stock assessment than the error in estimating the carbon content of peat soils, particularly deep peats for Meallmore.

Table 2.4. Predicted values of soil area and C stocks (kt C) to 1 m depth from the Scottish National (SN) Soil Map Classification (1:250K) and higher resolution Forestry Commission (FC) soil maps for two forest areas: Meallmore (M'more) and Craik.

Total SOC content (kt C)								
	Mineral Soils		**Peaty Soils**		**Deep Peats**		**Total**	
Mapping method	M'more	Craik	M'more	Craik	M'more	Craik	M'more	Craik
SN Soil Map	2	113	296	637	21	483	320	1233
FC Soil Maps	4	171	267	518	13	371	283	1060
Difference	-1	-58	29	119	9	112	37	173
% error*	-43.8	-51.7	9.8	18.6	40.3	3.3	11.4	14.0

* difference as % of Scottish National Map estimate.

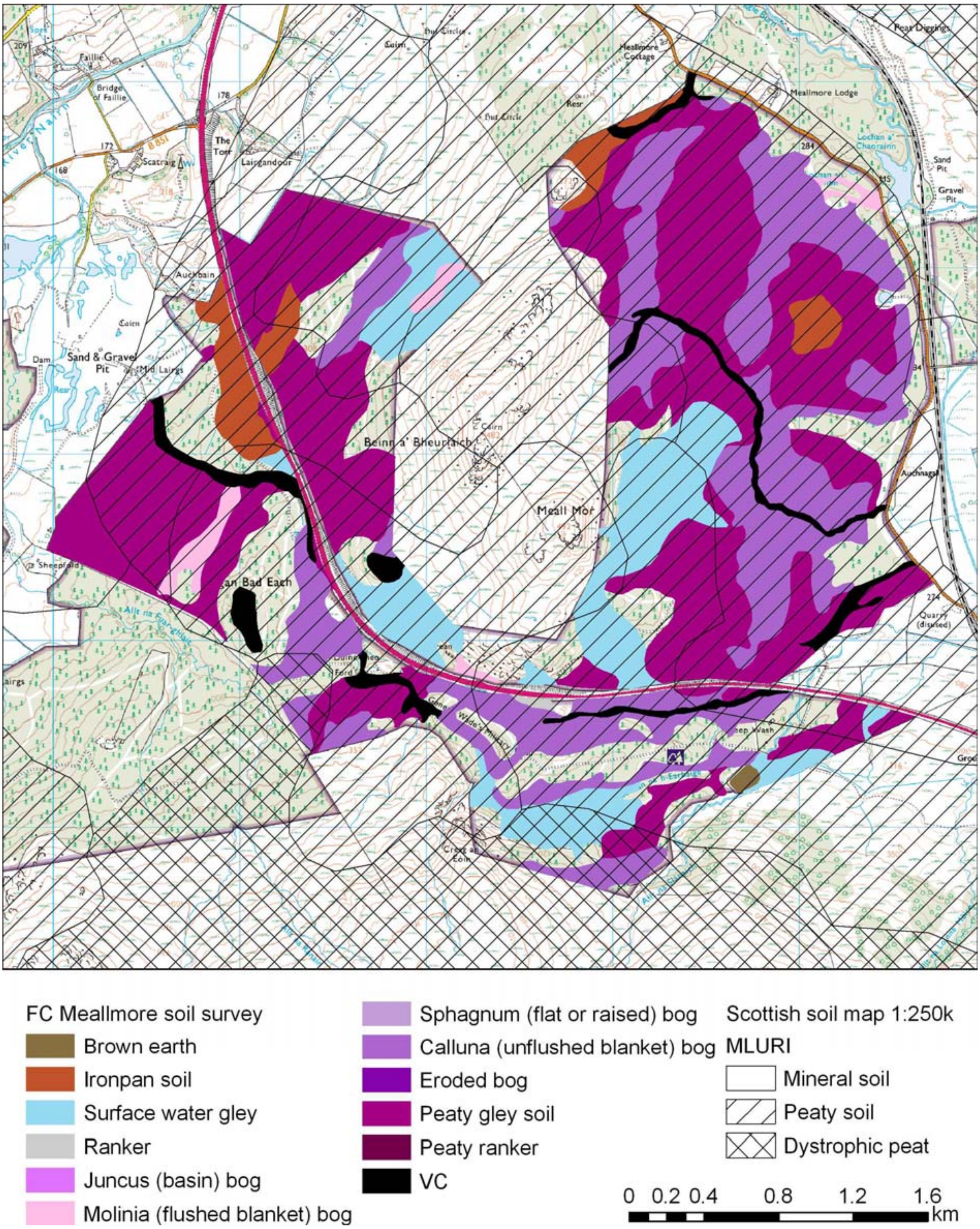


Figure 2.4a. A comparison between the MLURI Scottish soil map and FC soil survey map at Meallmore, Scotland. 'peaty' soils are shallow peats; 'dystrophic' are deep peats.

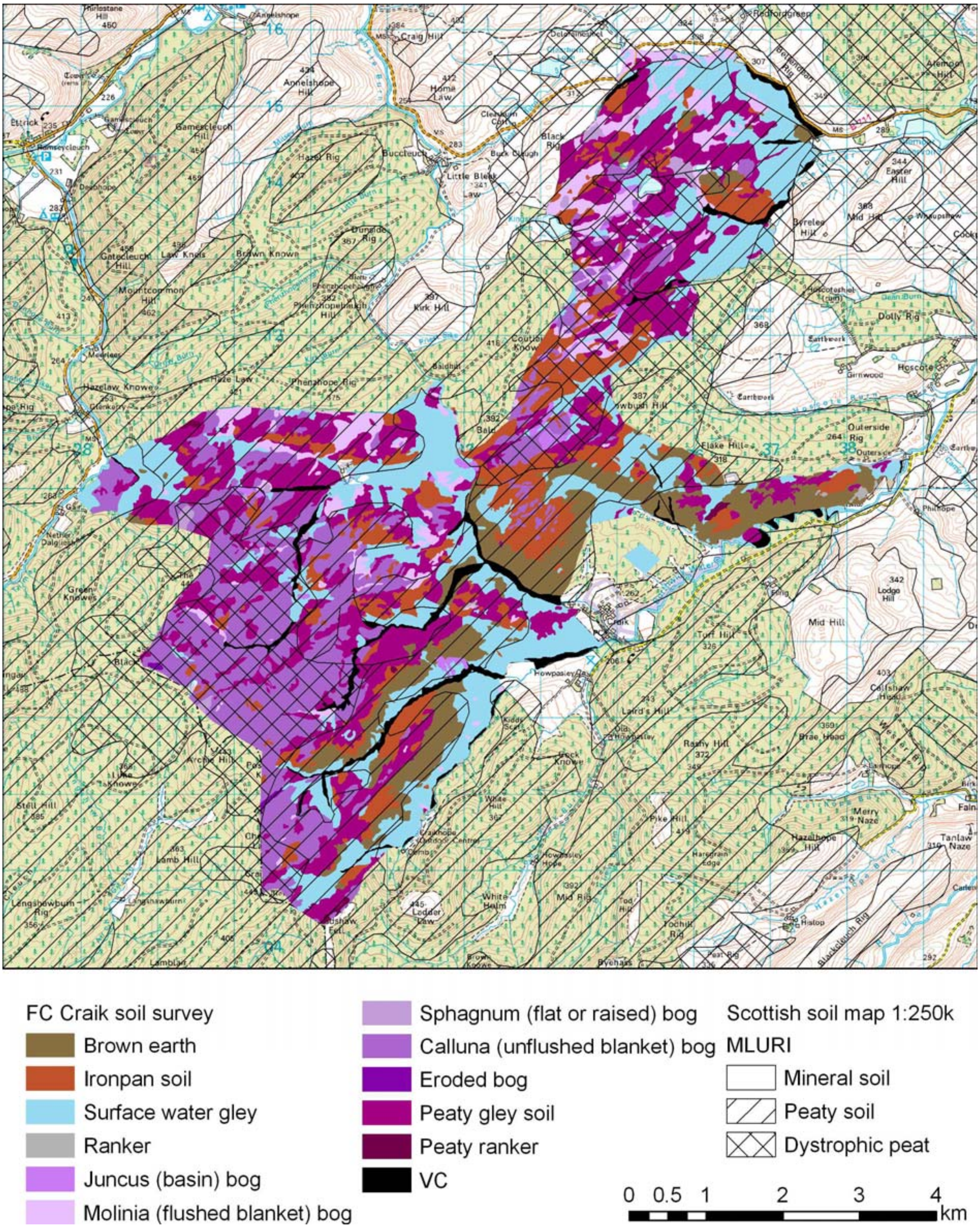


Figure 2.4b. A comparison between the MLURI Scottish soil map and FC soil survey map at Craik, Scotland. 'peaty' soils are shallow peats; 'dystrophic' are deep peats.

These overall errors are comparable to an 8% error estimated for Glensaugh catchment with predominately peats and podzol soils, in North East Scotland, by comparing measured soil C stocks with a National Soil Map (NatMap) UK soil database, reported by Frogbrook et al. (2009). That study was based on 300 ha (with 8% error) which is a considerably smaller area compared to 1009 ha (with 11% error) at Meallmore and 3881 ha (with 14% error) at Craik for our study. This suggests that the error may increase with the increase in area assessed. However, the associated error in soil C stocks at the Plynlimon catchment in mid Wales in the same study was 45% (Frogbrook et al., 2009), suggesting that much higher uncertainties could be associated with soil C national upscaling depending on the area, soil type and its spatial variability. **There is a need to test other locations, where there are a range of organic and organo-mineral soils encompassing other land use types and topographic situations, to contribute to improving the estimates of the amount of carbon held in Scottish soils.**

2.6 SOC components, stability and retention

Carbon can be lost from soils through:

- gaseous CO_2 emission during decomposition of organic matter (OM),
- methane (CH_4) emission,
- removal of soil particulate organic carbon (POC) in erosion,
- and loss in solution (in particular, dissolved organic carbon, DOC).

What is key in determining the rate of loss is the relative stability of the organic carbon, which depends on its form and where in the profile it is held.

Soil carbon stability

Soils contain different OM fractions with varying stability, turnover time and temperature sensitivity. Specific OM fractions are known to be more vulnerable to climate change and disturbances than others. Free Particulate Organic matter (FPOM) could be very high in the organic and top mineral soil horizons while Mineral Associated Organic Matter (MAOM) and Occluded Particulate Organic Matter (OPOM) are generally high in the mineral soil. Compared with the quicker turnover of FPOM, the slower turnover of OPOM is attributed to chemical recalcitrance, humification and physical stabilization by occlusion. MAOM is the dominant fraction in mineral horizons and has very slow turnover rates because of stabilisation by interaction with mineral surfaces, and iron/aluminium (Fe/Al) oxides and hydroxides.

Under coniferous forest, peaty soils and soils with a thick organic layer such as podzols may accumulate large amounts of OM in the organic layer within decades, making them vulnerable to climate change impacts and other disturbances. For example, mean turnover times of 4, 9 and 133 years, respectively, were calculated for FPOM fraction of Oi, Oe and Oa horizons by using a non-steady-state model in a podzol under coniferous forest (Schulze et al., 2009). **The faster turnover time for the organic horizons suggest the potential of these horizons to respond rapidly to changes in temperature or rainfall, and disturbance.** The turnover time for the MAOM fraction in mineral soils was between 390-710 years suggesting little contribution to the accumulation of soil carbon in these layers (Schulze et al., 2009).

In Table 2.5, the mean residence times (MRT) of soil C in the topsoil and subsoil under coniferous and broadleaved woodland reported in the literature are summarised. Topsoil C has a lower residence time because it has a high FPOM fraction. Subsoil C, mostly in mineral soils, has much longer residence times because it is composed mainly of MAOM which are much more stable soil C associations. Temperature has a strong influence on the turnover of soil C as suggested by much longer residence time in the Russian forest subsoil shown in Table 2.5 (Brovkin et al., 2008).

Table 2.5. Mean residence time (MRT) of soil C in the topsoil or subsoil under coniferous and broadleaved woodland reported in the literature.

Forest & Location	**Soil depth**	**MRT range (years)**	**Reference**
Spruce on Podzol Germany	Organic topsoil	*4-9*	Schulze et al., 2009
24 broadleaf woodlands UK	0-15 cm	*15-78*	Tipping et al., 2010
Broadleaf forest USA	0-30 cm	*11-31*	Garten & Hanson, 2006
Spruce on Podzol Germany	Top mineral soil	*133*	Schulze et al., 2009
Spruce/Larch Mainland Europe	Whole soil profile	*76-158*	Hahn & Buchman 2004
Surface forest soils European Russia	Surface forest soils	*130-625*	Brovkin et al., 2008
Spruce on Podzol Germany	Subsoil mineral soil	*390-710*	Schulze et al., 2009

The prevalent paradigm is that all soils contain fast, slow and passive forms of soil carbon (e.g. Tipping et al., 2009) with different residence times (typically 1, 20 and 1000 years, approximately), but in different proportions. Tipping et al., (2009) suggested a distribution of soil C pools between slow and passive in upland and lowland grassland and in broadleaved and coniferous forest (O layer) as shown in Table 2.6. According to this classification, soils under spruce forest have the least passive pool compared to others. This is likely to be due to high litter input, slow decomposition due to acidic conditions and accumulation of organic carbon in the humus layer which contain much higher proportion of the FPOM soil C fraction.

Table 2.6. Fractions of slow and passive SOM in soil under grassland and forestry (as estimated by Tipping et al., 2009)

Vegetation type	**slow**	**passive**
spruce forest O layer	0.84	0.16
deciduous forest	0.68	0.32
lowland grasslands	0.55	0.45
upland grasslands	0.29	0.71

Disturbance and retention of SOM

As shown above, organic C can be relatively stable, particularly that held at depth, remaining stored for several hundreds of years (McDowell and Likens, 1988). However, processes that disturb OM stability such as site preparation for planting, clear felling, destumping, etc., will shorten the turnover time of this stored carbon. Other factors governing OM and the retention of DOC in mineral horizons include pH and texture, particularly the proportion of silt and clay particles and the formation of soil aggregates, which protects OM from decomposition (Jones and Donnelly, 2004). Different soil types have different sensitivities to C release and different capacity to retain C in lower depths. Removal (adsorption) of DOC from the soil solution occurs via organo-mineral interactions on the surfaces of Al and Fe oxides and hydroxides in the mineral horizons as part of the podzolisation process. Dawson and Smith (2007) reported a study of soil solution DOC retention from a topographic sequence of upland soils in NE Scotland, and showed that compared to other upland soils, **peaty podzols had the greatest potential for DOC retention,** mainly by dissolution/precipitation reactions. Shorter contact time between the DOC and amorphous Al and Fe, an increase in soil solution pH and increased frequency of wetting drying and freeze-thaw cycles, all caused decreases in DOC retention in the Fe-rich horizons of podzols. **In highly organic soils the reduction in soil solution DOC concentration occurring with depth is smaller**, due to the lack of available mineral binding capacity.

Due to the soil OM origin, fate and composition, forest floors are more likely to lose carbon than underlying mineral soil layers when disturbed. For example, results from an extensive metadata analysis of the effect of harvesting (database of 432 studies from temperate forests, Nave et al., 2010) showed forest floor C storage to decline by a remarkably consistent 30 +/- 6 % (95% CI). In contrast, mineral soil layers showed no significant overall change in C storage due to harvesting. Losses in the organic layers were significantly smaller in coniferous/mixed stands (-20%) than hardwood (-36%) soils (Nave et al., 2010), presumably reflecting OM composition and environmental condition differences. This study, despite covering only organo-mineral soils, highlights the much higher sensitivity of the organic soil layer and organic matter to disturbances than mineral soil. If we translate this to organic soils, it **implies that deep organic soils have higher likelihood to lose more C due to disturbance compared with organo-mineral soils such as peaty gleys and peaty podzols**.

On the positive side, Nave et al. (2010) concluded that C losses could be mitigated, or even prevented, through the use of management practices that minimize physical disturbance of the soil profile. An important finding was that most of the 432 studies did not sample residues such as coarse woody debris as a component of the forest floor. Therefore, while forest floor C stocks did decline significantly, harvesting presumably increased the amount of C stored in woody debris pools.

<u>Measured DOC fluxes</u>

DOC is released into water moving through the upper organic horizons from the OM decomposition and mineralization in the soil promoted by microbial and fungal activity. Rates of loss will obviously depend on SOC amounts and forms, which will be influenced by vegetation types, as indicated above. A review by Hope et al., (1994), concluded that temperate forest ecosystems export slightly less DOC in rivers (121 kg CO_2 ha^{-1} y^{-1}) than moorland and grassland (157 kg CO_2 ha^{-1} y^{-1}). Michalzik et al. (2000) reviewed results for forest soils, and reported DOC fluxes for the Oa (organic) layer to range from 367 to 1470 kg CO_2 ha^{-1} y^{-1}, similar to the values calculated for 8 UK Level II forest sites (mean = 340, s.d. = 120 kg CO_2 ha^{-1} y^{-1} (Vanguelova et al., 2010). In their review of carbon loss from UK soil, Dawson and Smith (2007) reported DOC fluxes in a similar range (290 to 950 kg CO_2 ha^{-1} y^{-1}) for catchments of 24 upland streams and rivers, with a mean of 367 kg CO_2 ha^{-1} y^{-1}. Note that a DOC loss rate of 1000 kg CO_2 ha^{-1} y^{-1} is equivalent to a C flux of 0.272 t C ha^{-1} y^{-1} which is approximately only 0.08% per year of the C content down to 80 cm in a peaty gley soil, and approximately 5-10% of the typical annual C accumulation rate in conifer stands, averaged over a rotation. **Therefore DOC fluxes are relatively small components of the C balance in most cases.**

Many of the locations in the Dawson & Smith (2007) review contain substantial areas of peatland, and so the quoted surface water fluxes are more similar to DOC fluxes measured at Level II forest sites on highly peaty soil types such as Llyn Brianne and Coalburn. However, it should be noted that Buckingham et al., (2008) found higher DOC flux in the topsoil of moorland compared to topsoil of forest ecosystems. More generally, stream DOC fluxes are lower than topsoil values, due to the sorption and mineralization of DOC during transport through deeper soil (Kalbitz et al., 2000). It has been shown that hydrologic variability in soil horizons can be as influential as biological activity in determining DOC concentrations in soil solution (Kalbitz et al., 2000).

Interactions of soil carbon and nitrogen

The soil nitrogen content (N) is highly positively correlated with SOM content, **so the C amount is very important in determining the fate of N in the soils and *vice versa*.** Soil N stocks were also estimated alongside soil C stocks from the 167 BioSoil sites. The total N content up to 100 cm soil depth in forest soils followed the order of soil C content, and across GB ranged between 7 and 24 t N ha^{-1}. Both C and N soil stocks across the different soil types decreased in the order deep peats > peaty gleys > surface water gleys = ground water gleys > brown earths = podzols. The C:N ratio in the forest floor gives an indication of availability of nitrogen in floor material and its rate of decay (Emmett, 2007). The majority of forest soils have bulk C:N ratios between 20 and 40, although C:N ratios above 40 in organic layers frequently occur in Northern Europe, which is mainly due to low N input and slow organic matter decomposition rate. Values between 20 and 60 have been measured in litter layers of shallow and deep peat soils in Scotland, The C:N ratio decreased and the litter N concentration increased from north to south along an increasing N pollution gradient (see Figure 2.5; Vanguelova et al., in prep.). This suggests that in northern Scotland, peat decomposition is slow and N deposition low, while in southern Scotland some forest ecosystems with a forest floor C:N ratio of <25 may show mineralisation of N and leaching to ground water. This also means that **peat soils in the south of Scotland experience higher N deposition and are likely to have higher decomposition rates and be more sensitive to disturbance**.

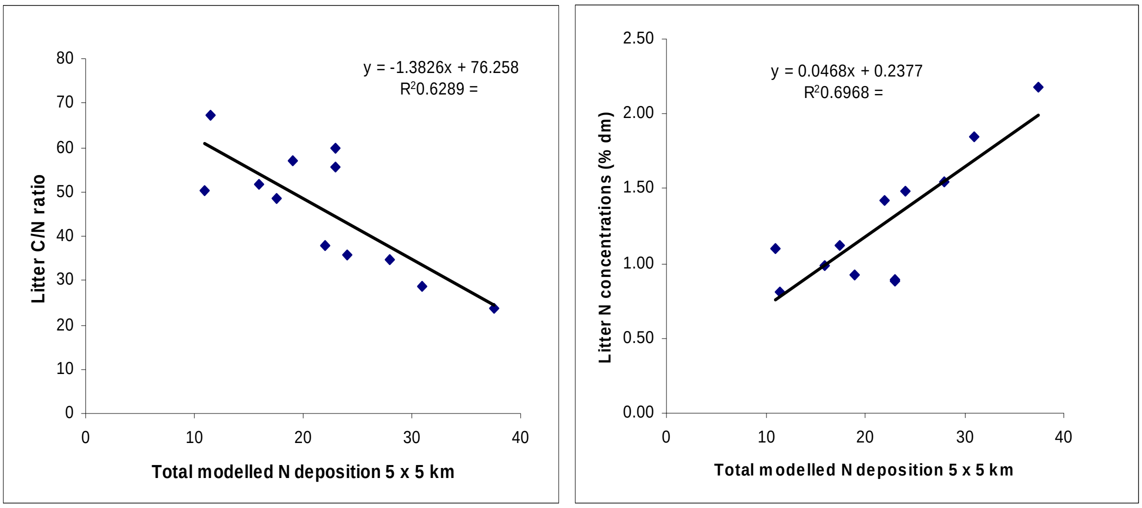


Figure 2.5. Relationships in forested sites in Scotland between total modelled N deposition and (a) litter C:N ratio and (b) litter N concentration deep peat soils. Data taken from the BioSoil network (Vanguelova and Pitman, 2009).

3. GHG emissions and C stock changes associated with forestry on peaty soils

3.1 GHG emissions from peat soils

GHG fluxes from soil, which include carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are affected by many natural factors (e.g. soil moisture content, water depth, aeration, soil type, soil temperature, and soil pH) and man-made factors (such as land use change and management). There are also strong interactions between these factors, for example through N input and soil C:N ratio. **Whilst C:N ratio is the most important factor affecting between-site N_2O spatial variation, the groundwater table depth may be the most important factor determining the size of soil CO_2 and CH_4 fluxes** (von Arnold et al., 2005a). For example, Freeman et al. (1993) simulated, experimentally, the effect of climate change-induced water table draw-down using undisturbed intact monoliths from a Welsh peatland. Lowering the water table decreased CH_4 efflux (maximum -80%) and increased both CO_2 and N_2O efflux (maximum 146% and 936%, respectively). Production of CH_4 is strictly anaerobic, production of CO_2 aerobic and N_2O can be produced under both aerobic and anaerobic conditions and may be consumed by wet, nitrogen-poor soils (Chapuis-Lardy et al., 2007). The amount of oxygen in the soil (determined by the level of the water table), and soil temperature are thought to be the key influences on the production and consumption of these gases in peat soils. Oxygen availability therefore means that the top 20 cm layer of the soil dominates the net GHG fluxes, although CH_4 production can occur at depth and can reach the surface through cracks. Clearly, **disturbance during ground preparation for planting or restocking is likely to affect soil water and aeration conditions and thus GHG fluxes. However, there is very little directly relevant data**.

An additional difficulty is that simultaneous measurements of all GHGs are necessary to calculate the CO_2-equivalent flux in order to estimate the total Global Warming Potential (GWP, see Glossary for explanation) of forestry, forestry practices and any land use change activities. **There are very few measurements of simultaneous CO_2, CH_4 and N_2O fluxes either outside the UK** (von Arnold et al., 2005a,b,c; Minkkinen et al., 2002) **or within the UK** (Zerva and Mencuccini, 2005; Freeman et al., 1993). Those reported are measured by chamber methods and therefore have large uncertainties due to the difficulty in capturing the large spatial variation. There are also large temporal variations. For example Ball et al. (2007) measured annual CH_4 emissions from a peaty clearfelled forest area in Harwood, England and found large inter-annual variations with fluxes of 0.17 ±0.03 and 0.45 ±0.03 t CO_2eq ha^{-1} y^{-1} (compared to CO_2 emissions of 24-26 t CO_2 ha^{-1} y^{-1}).

The available data on soil GHG emissions associated with forestry and from peatlands in the UK published between 1990 and 2009 are summarized in the table in Appendix B (positive numbers indicate net emission), with information on sites, tree species and or vegetation type. Values for GHG emissions from other vegetation on peat soils are also given for comparison. There are obviously many ways in which these fluxes could be categorized and presented (e.g. by soil type, tree species, climate, methodology etc.) and ideally the categorisation could identify the uncertainties involved. However, because of the paucity of relevant data for all GHGs, and the large variations and uncertainties within the fluxes the table does not distinguish between peats of >50 cm depth or shallow peats (<50 cm). The results have been grouped based on standing forest, clearfell or other vegetation. The last category highlights the point emphasised in the comprehensive report by Lindsay (2010) that there is almost no GHG flux data for the UK or comparable areas for genuinely undisturbed peatland.

The total CO_2 equivalent emission rate in the table in Appendix B was calculated by multiplying the median emission rate of each gas by its GWP and summing. The ranges of soil GHG emission rates (and the median value) in the 3 categories are shown in Table 3.1. Differences between categories should not be overemphasised due to the incomplete data and lack of direct comparisons. However, it is clear that the **CO_2 flux from the soil dominates in the total GWP, although CH_4 fluxes can be substantial in the wettest sites** (Appendix B). The total GWP (expressed as t CO_2eq ha^{-1} y^{-1}) reduced in the order: Other vegetation > Clearfelled > Standing forest.

Table 3.1 Summary of soil GHG emission rates measured in UK forests and peatlands sites. Values are ranges with median in parentheses; negative values indicates uptake. Note the differences in units for CH_4 and N_2O.

Peatland type	**CO_2** t CO_2 ha^{-1} y^{-1}	**CH_4** kg CH_4 ha^{-1} y^{-1}	**N_2O** kg N_2O ha^{-1} y^{-1}	**Total CO_2eq** t CO_2eq ha^{-1} y^{-1}
Forest	3.7 - 340.8 (17.5)	-9.1 - 2.8 (-0.51)	0.1 - 4.7 (0.56)	16.8
Clearfelled	5.1 - 26.0 (11.8)	6.8 - 18.0 (8.8)	0.7 - 51.0 (2.5)	21.3
Other vegetation	21 - 55.8 (25.4)	0.2 - 53.3 (1.8)	-0.1 - 1.86 (0.3)	31.6

The recent study of Dinsmore et al. (2009) from Auchencorth Moss, Scotland (mosaic of Calluna, grasses, sedges and riparian rushes, ~85% peat), highlighted the underlying environmental and vegetation characteristics which lead to within-site variability in both CH_4 and N_2O emissions and the importance of such variability in up-scaling. They indicated that CH_4 emissions varied considerably across the catchment, and appeared to be linked to areas with consistently near-surface water tables with the riparian zone representing a significant hotspot (Appendix B). Contrary to many previous studies, the presence of either sedges or rushes containing aerenchymous tissue decreased net CH_4 emissions during the 2 growing seasons. Upscaling the calculated fluxes using vegetation cover estimates from a satellite image, gave annual mean (and coefficient of variation) catchment CH_4 and N_2O emissions of 1.06 kg ha^{-1} yr^{-1} (300%) and 0.02 kg ha^{-1} yr^{-1} (410%) respectively, although these values are sensitive to error in the cover estimates. These GHG fluxes are very small compared with the CO_2 flux of between 20 and 40 t CO_2 ha^{-1} y^{-1} , when compared as CO_2 equivalents: CH_4 = 0.03 t CO_2eq ha^{-1} y^{-1}, N_2O <0.001 t CO_2eq ha^{-1} y^{-1}.

3.2 GHG emissions from peatland vegetation

At the stand-scale (*i.e.* soil and vegetation cover) a comprehensive review of carbon and GHG fluxes from European peat bogs and fens comes from Byrne et al. (2004). Table 3.2 shows the emission rates for GHGs based on median values of all the reviewed data for ombrotrophic bog and minerotropic fen sites, both 'undisturbed' and managed. In pristine bogs CH_4 fluxes are very large and important components of the GHG balance, although very variable spatially and temporally. In managed peatlands the CH_4 fluxes are considerably smaller. N_2O fluxes are only substantial components of the GHG balance in peatlands managed for arable crops and nitrogen fertilisers applied. The study concluded that when emission rates are summed as CO_2eq ha^{-1} y^{-1} these types of peatland generally are sources of GHGs with emission intensities increasing in order:

> bog: forestry < undisturbed < restoration < new drainage for forest/peat cut < peat cut < abandoned after harvest = grass < crop;
> fen: (restoration <) forestry <= undisturbed < new drainage for forest < grass < crop.

Afforested bog is the only category in which there is a negative value for CO_2eq, because of the net CO_2 uptake, and the low CH_4 emissions. However, Byrne et al. (2004) noted that although afforested peatlands emit less GHGs than undisturbed bog or mire caution in interpretation is required because the studies cited deployed mild drainage only, at which CH_4 emissions are reduced but peat formation may still continue. They emphasised that emissions vary with stand age as transpiration by the trees affects the water table leading to high CH_4 emissions under young stands and significant drainage and therefore larger CO_2 emissions under older stands. Furthermore, they point out that the long time-scale for peat soil C balance to come into equilibrium with altered conditions, compared to the vegetation.

Table 3.2 Stand-scale emission rates (median values) based on measured fluxes from different bog and fen management types in European peatlands (recalculated from Byrne et al. 2004). Negative values indicates uptake. Note the differences in units for CH_4 and N_2O.

Peatland Type	**CO_2** t CO_2 ha^{-1} y^{-1}	**CH_4** kg CH_4 ha^{-1} y^{-1}	**N_2O** kg N_2O ha^{-1} y^{-1}	**Total CO_2eq** t CO_2eq ha^{-1} y^{-1}
Bog (ombrotrophic)				
Afforested (drained)	-0.7	14.9	0.06	-0.31
Drained (for forest & peat cut)	4.03	26.7	0.04	4.72
Grassland	8.62	2.7	0.02	8.69
Arable	16.1	0	0	16.1
Extracted (peat cut)	6.42	23.0	0.63	7.18
Restored	2.27	20.0	0.03	2.78
Pristine (temperate)	-2.6	232	-0.02	3.19
Pristine (boreal/sub-arctic)	-0.73	50.0	0	0.52
Fen (minerotrophic)				
Afforested (drained)	-0.73	-0.07	2.88	0.12
Drained (for forest)	1.47	1.33	1.65	1.99
Grassland	15.1	0.53	7.94	17.5
Arable	15.0	-0.27	18.2	20.4
Restored	No data	16.5	1.01	0.71
Pristine (temperate)	-1.47	189	No data	3.27
Pristine (boreal/sub-arctic)	-1.80	160	0.18	2.26

3.3 Implications of climate change

Changing climate variables (mainly rainfall, temperature and snow) may affect peat water table depth, surface runoff and vegetation species and have a direct effect on GHG fluxes. In Scotland, climate projections are for warmer conditions (warmer summers, milder winters), and more seasonality in rainfall (drier summers, particularly in the East, UKCIP, 2009). A higher frequency of very dry summers is also projected for eastern Scotland, and generally more high rainfall events. Such **changes in the climate will inevitably affect GHG emissions from existing peaty soils, as well as changing the conditions for their formation.** High rainfall events may cause more water-logging, and runoff of POC and DOC, as well as higher water tables and increased CH_4 emissions. Warmer conditions will extend the growing season and the period of more active soil decomposition and the effect may be relatively large in cooler upland areas. Drier conditions will lower water tables, increasing CO_2 emissions, but reducing CH_4. Prolonged drought may cause or exacerbate peat cracking, and thus increased drainage and DOC & POC loss. When severe, cracking can permanently lower the water table and furthermore, will probably make rewetting for bog restoration impossible. Climate change may even cause more wild-fires, causing major pulses of CO_2 and NO_x emissions, at a local scale, and several major consequences for the C balance of peatlands. Kesik et al. (2006) modelled the effect of climate change on forest soil N_2O emissions across Europe, and for the UK predicted a 24% increase in N_2O emissions in UK & the Republic of Ireland. However, the results were not disaggregated for soil type, so of limited use for consideration of peat soils alone, and as discussed above with Table 3.2, N_2O emissions from peat soils are usually a minor component in the GHG balance.

The recent ECOSSE reports, (Smith et al., 2007, 2009) discuss the possible effects of climate change on peatland erosion and peat loss in detail. They conclude from their scenario modelling that climate change between 1990 and 2060 will result in a decline in Scottish soil C stocks of only 93 to 125 kt C, less than 0.01% of present C stocks, and nearly 1/50th of the likely changes from land use (Smith et al., 2009). However, their analysis does not take into account changes in vegetation composition driven by climate change which will also alter peat formation and protection.

3.4 Past and present forest establishment practices

Appropriate cultivation techniques for the preparation of soils for tree planting underpin the successful establishment of forestry tree seedlings and capture of site nutrients, especially when restocking of previously afforested land (Patterson & Mason, 1999). The effect of cultivation for tree stability has also had considerable research effort due to the windy conditions in upland areas of the UK. Cultivation of peaty soils will inevitably result in some increase in SOC loss, at least in the short term, although exact quantification is very difficult. Considering GHG abatement objectives in isolation, the key issue for forest managers is choosing appropriate sites, tree species and cultivation techniques taking into account SOC losses and likely overall stand C stocks gains under trees. These ideas are explored below, although it is recognised that there are likely to be other local objectives, not simply improved GHG balance, which will require consideration.

In the 'peak periods' of afforestation in upland Britain in the 1960's, 1970's and 1980's significant levels of site modification were often necessary using drainage, agricultural plough and tine, moling/ripping, repeat applications of fertiliser additions and control of weed growth. Current evidence and guidance identify that site characteristics, especially soil type and proposed woodland type or species should be considered, and recommendations for suitable cultivation methods across the major soil groups are available (Paterson and Mason, 1999). The research experience and available expert knowledge highlights that, **for all apart from the most fertile of sites, there are tree survival and growth benefits from some form of soil cultivation.** Where a site has been previously intensively cultivated it may be possible to utilise existing plough furrows at the time of restocking without further cultivation, although active brash management is likely to be essential. Direct planting of new planting sites is unlikely to result in a uniform spaced crop that exhibits high levels of survival and good early growth. Some species and soil combinations rule out direct planting because there is insufficient labile nutrient availability, e.g. spruce on podzols. Direct planting is, therefore, most commonly proposed when economic timber production is not an objective for the establishment of woodland cover on a site. Scarification is a good site cultivation approach where surface water is limited, and a minimal peat layer is present (e.g. shallow peaty podzol) although increased weed control measures will be required in order to achieve good establishment.

3.5 Establishment options on different peat soils

On soils with a peaty layer or deep peats concerns over nutrient run-off, stream eutrophication and soil C losses (particularly DOC & POC) when sites were completely cultivated by agricultural plough have led to changes in practice. **The most common site cultivation method currently deployed is some form of excavator mounding system.** This technique effectively provides the removal of competing vegetation, local site drainage to reduce water table depth and improved root temperature regime (Tabbush, 1988). If an ironpan or indurated layer is present in the soil profile, presenting an impediment to root growth and likely to reduce tree stability, then ripping is usually required, and is most commonly used in conjunction with scarification. Sites on deep peats where afforestation may be appropriate are those that have been previously degraded by peat extraction or agricultural modification (Patterson and Anderson, 2000). These are probably losing substantial amounts of SOC, and the net GHG balance is likely to be improved by growing trees (see Table 3.2).

The level of disturbance from normal afforestation and restock operations on high SOC soils is commonly affected by two main factors, [1] the type of cultivation prescribed, and [2] site drainage. For excavator-based techniques the variation in methods and bucket geometry are important determinants of the level of soil disturbance. Where soils in FC soil classification code 6 (Kennedy, 2002; peaty gley up to 25 cm peat layer and including code 6p with 25 to 45 cm peat layer) and deeper peats (only 9b and 9c) are considered for afforestation then careful prescription of the cultivation regime is required, in order to minimise SOC loss and maximise tree growth. When restocking a site with active drains on Type 9 soils an excavator screef or hinge cultivation will often be sufficient for establishment of Sitka spruce, with a nurse crop mixture.

As sites with high SOC are harvested at the end of the first rotation it is important to critically assess their potential for successful restocking, especially where productive conifer forest is a key objective. **On deep peats where tree growth is likely to be poor, substantial site modification through new drainage, cultivation or fertilization may be required to achieve good growth from commercial conifer species, such that increased GHG emissions and SOC losses lead to the potential for net GHG balance to be negative,** and other options should be considered. On deep peats (*i.e.* >50 cm depth), the site types that may provide an opportunity for conifer restocking are FC code 9b and 9c *Molinia flushed blanket* bogs with tussocks[2] (Kennedy, 2002, see also Appendix C), which are classed as 'poor nutrient status' (in the ESC system) or in 'Taylor class' B/C in guidance for fertilisation (Taylor, 1991). In such sites pine and/or spruce and pine mixtures are a pragmatic approach to establishment, with a likely requirement for P and K fertiliser addition in the early part of establishment. Fallow periods, after harvest on 9b & 9c sites, will incur nutritional losses and reduce second

[2] Note that the FC classification of deep peats uses the natural plant species communities present to define peat types (Kennedy, 2002); where these are not present in restock sites, Kennedy (2002, p. 40-41) gives an alternative key to be used in the field.

rotation growth. The other type of *Molinia* bog, code 9a, is very wet and most suited to leaving for rewetting through active management of natural processes and the development of bog woodland with Salix and other pioneer tree species such as birch.

Sites categorized as code 9d and 9e *Trichophorum* (deer grass) *bogs* are often weakly flushed and very wet or wet and 'poor or very poor nutritional status' (ESC system). On code 9d the presence of *Calluna* will further exacerbate the already poor nutritional status and lead to a likely requirement for repeat fertiliser additions for successful forest development. Therefore, code 9d and 9e sites should not be considered for conifer restocking and should be prioritized for open habitat restoration so that the continued loss of SOC will eventually be stopped, and this also applies to FC code 8 sites (*Juncus or basin* bogs), with the possible exception of 8c where inherent good soil nutrient status and low priority for restoration means second rotation establishment without a requirement of fertilisation is possible.

For FC code 10 sites (*sphagnum* or *flat* or *raised* bogs) restoration should be considered (Patterson and Anderson, 2000). Where category 11 bogs are encountered (*unflushed blanket bogs*), as with 9d sites, the *Calluna* reduces further the already 'very poor' (ESC) or C/D (Taylor) nutritional status. Thus the high requirements for N addition and heather control would suggest that restocking should be avoided. With the final category of deep peats *eroded bogs* (FC code 14) the likelihood is that the high water table will prevent successful machinery use and require motor-manual felling thus the most sensible choice will be open space management through limited intervention. It should be noted that the recent Natural England report on peatlands (Natural England, 2010) concluded that after felling, the restored bog vegetation would sequester C more slowly than the forest, so that initially, the restoration would be unlikely to deliver net GHG balance benefits. However, the C loss from the peat would be slowed, and successful restoration would deliver new long-term C sequestration (Natural England, 2010).

3.6 Consequences of different site preparation practices for GHG fluxes

Soil Disturbance

Although considerable evidence exists of the benefits of site cultivation for establishment and growth on the majority of upland soil types, **there are few data to help understand the impacts of soil disturbance on soil carbon stocks and GHG fluxes.** Johnson (1992) reviewed the literature on the effects of various forestry practices upon SOC. He found that site preparation in general led to carbon losses, which varied with the severity of the disturbance. The ground area disturbed by common site preparation treatments has been quantified by Worrell (1996). As expected, mechanised preparation disturbs more soil than manual planting and therefore it is logical to assume that ploughing is likely to cause the greatest carbon losses, whereas hand screefing is likely to cause the smallest carbon loss. However, levels of GHG emissions are likely to be modified by the amount of organic material either within the soil, in the case of afforestation, or by material left on the site following harvesting when restocking.

Investigations in Canada by Schmidt et al. (1996) and Mallik and Hu (1997), both found that mechanical site preparation could cause a significant loss of soil carbon (although one site in the investigation of Schmidt et al. (1996) showed no change in soil carbon content). However, a literature review and meta-analysis by Paul et al. (2002) concluded that there is no significant effect of disturbance level, and that instead any decrease in soil carbon is attributable to the reduced plant matter input to the soil in the early years of tree growth (< 10 years). Overall, the ECOSSE review (Smith et al., 2007) of organic soils concluded that **minimizing disturbance during forestry operations was very important to reduce the impact on soil C**.

Afforestation

It is clear from the previous sections that afforestation of peatland could have significant impacts on SOC loss due to drainage and cultivation. This may cause a shift from a carbon sink in the pristine peatland ecosystem, to a carbon source (Cannell et al., 1993; Worrall et al., 2010) depending on the net CO_2 uptake by the forest. The recent Natural England report (Natural England, 2010) concluded that most afforested peatlands are net sinks for GHG during early rapid growth, but are net sources later when forests mature. The ECOSSE review (Smith et al., 2007), concluded that “afforestation probably has little net effect on soil organic carbon stores in organo-mineral soils, but this statement is very uncertain” because of conflicting reports. In addition, the few investigations quantifying the SOC changes following afforestation tend to be restricted to comparison across chronosequences, rather than measurements taken through time. Worrall et al. (2010) assessed the available reports and concluded that there is a high probability that afforestation improves the C balance of deep peat soils, and a high probability that afforestation improves the overall GHG balance by reducing CH_4 fluxes as well.

Cumulative SOC loss from afforested peat over time has been estimated at about 20-25% of the C in the peat (Harrison et al., 1997; Jones et al., 2000) and some modelling analysis has assumed even higher losses over long time periods for deep peat (e.g. 750 t C ha^{-1}, Bateman and Lovett, 2000). However, there is great variability and uncertainty in such estimates (see e.g. Reynolds, 2007), in part due to varying time periods studied and assumptions about equilibrium conditions. Furthermore, these estimates were derived from sites with past ground preparation and drainage techniques which typically caused substantial peat disturbance and loss. It is not clear that such estimates are now appropriate because current planting and ground preparation practices are likely to lead to much lower changes in SOC. These issues emphasise the need for detailed quantification of SOC changes during afforestation.

Hargreaves et al. (2003) measured stand-scale CO_2 fluxes on deep peat sites in Scotland following ploughing for afforestation and found one site became a net C source in years 1 and 2 after planting, peaking with a net emission of approximately 14 t CO_2 ha^{-1} y^{-1} at year 2, prior to establishment of much vegetation. A similar site 8 and 9 years after planting was a net C sink with a maximum value of approx. 7.3 t CO_2 ha^{-1} y^{-1}. However, this data was only collected over short periods, from different sites, and does not agree well with other studies, although few examine soil+vegetation C balances. For example, Byrne and Farrell (2005) studied the effect of afforestation on soil CO_2 emissions from undrained blanket peat sites in Ireland across a chronosequence that had been drained and established at 3, 19, 23, 27, 33, and 39 years previously. Generally, the average annual soil CO_2 emissions showed no clear differences between the different sites (ages) and the authors suggested that afforestation does not always lead to an increase in soil CO_2 emissions. They also found that soil CO_2 emissions at the most recently forested site were approximately 6.2 t CO_2 ha^{-1} y^{-1} where drainage had not lowered the water-table, less than half the peak emissions value found by Hargreaves et al. (2003) shortly after planting. Zerva et al. (2005b) looked at the effects of afforestation of a peaty gley site in Northern England upon soil C balances, and found differences between first and second tree crop rotations. Whilst there was no detailed information on fluxes in the first rotation following afforestation, they inferred from the 73% higher soil C content in the H layer of unplanted moorland compared to afforested areas that the first 40-year rotation resulted in a decrease in SOC equivalent to 12.5 t CO_2 ha^{-1} y^{-1}. They attributed this decline to the accelerated decomposition caused by the site preparation of deep ploughing and drainage (see also Jarvis et al., 2009). Subsequently, in the second rotation there was a recovery of SOC (see below, Restocking).

A re-analysis of four peatland afforestation studies by Reynolds (2007) of different ages found that soil C *increased* on average 1.8 t CO_2 ha^{-1} y^{-1}, although there was considerable variation (0.7-2.1 t CO_2 ha^{-1} y^{-1}). Reynolds (2007) also emphasised the importance of other C losses through stream water, as dissolved (DOC) or particulate organic matter (POC) (as discussed in Section 2.5).

Harrison et al. (1995) quantified values of potential soil C losses that may occur due to drainage in relevant literature from around the world. Values ranged from 1.5 - 18.3 t CO_2 ha^{-1} y^{-1}, depending on either mean temperature or a combination of temperature and annual precipitation. Afforestation and drainage lower the water level and may thereby increase decomposition. However, CO_2 emissions follow an optimum function, with highest rates at intermediate water table levels (Glatzel et al., 2006). Drainage may also increase nutrient mineralization which can lead to increased N_2O emissions. Application of fertilizer at early stages of afforestation (or high atmospheric N deposition) helps accelerate the decomposition of soil OM increasing CO_2 emissions, although the application of fertilisation is currently limited in the UK. However**, any drainage induced increase in soil CO_2 efflux would result in a concomitant reduction in CH_4 emissions from the soil** (Cannell et al., 1993; see section 3.1). Nykänen et al. (1998) indicated that lowering of the water table by 10 cm would induce a 70% reduction in emissions from fens and a 45% reduction from bogs. Studies in boreal peatland have shown that CH_4 emissions decreased (or the soil became a net sink) following drainage and afforestation (e.g. von Arnold et al., 2005c and Maljanen et al., 2003).

Mojeremane et al. (2010, see Table 3.3) studied the effects of site preparation for afforestation (effect of drainage, mounding and fertiliser application) on CH_4 fluxes from grassland on a peaty gley soil at Harwood Forest in NE England. They found that the overall soil CH_4 emissions were significantly decreased by drainage (-64%) but increased by mounding, and fertilization (+41% and +44%, respectively). Note that in this case planting preparation treatments still remained net sources of CH_4.

Table 3.3 Mean CH_4 fluxes (also expressed as t CO_2eq ha^{-1} y^{-1}) from the different site preparation treatments at Harwood Forest, measured over 2 years, from Mojeremane et al. (2010).

Treatment	**CH_4** kg CH_4 ha^{-1} yr^{-1}	**CO_2eq** t CO_2eq ha^{-1} y^{-1}
Drained	6.3	0.16
Undrained	17.6	0.44
Mounded	14.0	0.35
Unmounded	9.9	0.25
Fertilised	14.1	0.35
Unfertilised	9.8	0.25

Finally, in considering the overall GHG balance during afforestation, it should be recognised that afforestation may result in the removal of significant amounts of existing vegetation, either rapidly during ground preparation and planting, or slowly as the trees grow and suppress existing vegetation (or a mixture of both of these). Typical peatland semi-natural vegetation biomass is between 2-22 tCO_2 ha^{-1}. In heathland ecosystems with diverse scrub communities higher C stocks are likely to be present in the vegetation.

Thus to make net C stock gains, the tree growth must exceed this amount; which is likely within a few years for productive sites and tree species combinations.

Clearfell and Restocking

Restocking (*i.e.* the second forest rotation) on a peaty gley soil in the UK resulted in an *increase* in soil C at a rate of approximately 14.6 t CO_2 ha^{-1} y^{-1}, assessed after the first 12 years (Zerva et al., 2005), although these estimates have considerable uncertainty (*cf.* Conen et al., 2004). A key point made by Jarvis et al. (2009) about this case study was that **the recovery of SOC after restock depended on minimising disturbance and on leaving harvesting residues on site**. This is probably attributable to restock excavator mounding burying surface OM and prolonging residence times. On shallower peaty gley soils, mounding may also mix peat with mineral layers, though the effect on decomposition, SOC stability and soil C loss is currently unknown. Zerva and Mencuccini (2005b) reported soil CO_2 emissions on a peaty gley site at Harwood in the first 10 months after clearfelling Sitka spruce of 15.1 CO_2 ha^{-1} (equivalent to 18.1 t CO_2 ha^{-1} y^{-1}), 71% of that in a nearby 40 year-old stand. Zerva et al. (2005) reported soil CO_2 emissions of 20.5 t CO_2 ha^{-1} y^{-1} for a similar site 18 months after clearfell and standard mounding preparation, while Ball et al. (2007) reported a soil CO_2 efflux of 26.0 and 23.7 t CO_2 ha^{-1} y^{-1} for two different years starting 18 months after clearfell. At that site stand scale CO_2 flux measurements were only 4.0 t CO_2 ha^{-1} y^{-1} (Kowalski et al., 2004), indicating substantial CO_2 uptake by vegetation re-establishing after preparation. The estimated total ecosystem respiration value for the site was 40.4 t CO_2 ha^{-1} y^{-1} thus Ball et al. (2007) estimated that the losses from decomposition of woody residues (brash, stumps) retained on the clearfell site were about 14.4 t CO_2 ha^{-1} y^{-1} (Ball et al., 2007).

Several of these authors highlight the importance of water table depth on respiratory losses of CO_2 from the soil. In addition, some contributions to total ecosystem respiration are likely to come from ectomycorrhizal decay, which is often 'missed' by static chamber assessments. Heinemeyer et al. (2007) suggest this contribution could be around 25% in a growing pine forest, and the ectomycorrhizal contribution to immediate post-harvest soil efflux is likely to be large but short-lived. Other UK studies show that the net CO_2 uptake of second rotation Sitka spruce stands range from 25.6 t CO_2 ha^{-1} y^{-1} at canopy closure to 11.0 t CO_2 ha^{-1} y^{-1} in older stands (Kowalski et al., 2004; Magnani et al., 2007). Thus it appears the increase in total below-ground C allocation with stand age and later rotations, which is proportional to tree productivity, accounts for the gains in SOC exceeding the rate of C loss from the soil during clearfell and restocking. Indeed, in a recent study from Finland of previously agriculturally cropped organic soils, Mäkiranta et al. (2008) observed that afforestation *reduced* peat decomposition rates. Afforestation of those peatlands abandoned from cultivation or peat harvesting in Finland can reduce total CO_2eq emissions and reduce the C lost from the soil (Alm et al., 2007).

Emissions from forestry operations

The CO_2 emissions from fossil fuel combustion associated with seedling production, machinery use etc. is fairly consistent across site types. Excavator use in site cultivation is estimated to emit 0.655 t CO_2eq ha^{-1} whilst machinery emissions for scarification average 0.238 t CO_2eq ha^{-1}. Ploughing of agricultural land for afforestation gives average calculated emissions of 0.073 t CO_2eq ha^{-1} (see Mason, Nicoll & Perks, 2009, Table 6.8). Note that these machinery emissions are "one-off", not continual or repeated. Therefore **emissions from forestry operations are very small compared to potential continuing SOC losses from enhanced peat decomposition noted above, and very small in comparison to the potential medium to long term C uptake during tree growth**.

3.7 Developing a predictive understanding of forest GHG balance on peat soils

Overall, it is important to acknowledge that the observed and reported changes in GHG fluxes and C stocks for forestry on peat soils are from a limited number of investigations, and also only provide values, in the main, for the early part of the rotation. Later changes during the rotation should also be considered so that an integrated, long-term forest GHG balance accounting for various management options, rotation lengths and rotation cycles can be considered.

The application of a gap-type model simulating the dynamics of forest ecosystems in Finland (Pussinen et al., 2002) suggested that the soil C stock in the organic layers did not increase as the length of the rotation period increased from 40 to 110 years. The amount of SOM was at its maximum about 5 years after the clearfell, at the time when the input from litter was high due to the decomposition phase of the logging residues. Therefore, the amount of SOM was highest when the rotation length was relatively short causing maximum woody residue inputs to the ecosystem (Pussinen et al., 2002).

For upland forestry in the UK it is therefore difficult to quantify the long-term net benefit of afforesting or restocking a site. The UK results discussed above provide a series of 'snapshots', measurements at particular time points. However, in order to answer questions of 'net benefit' additional information is required on stocks and fluxes, and a modelling synthesis of the existing data so that impacts can be assessed for the predominant forest types.

A process-based model of tree growth has been developed that uses data from the few existing studies of site CO_2 exchange and allows parameterisation of tree growth aspects from mensurational datasets to allow predictions of forest growth. The hybrid process-based tree growth model 3PG, coupled to a soil carbon-nitrogen sub-model ICBM to create 3PGN, has now been applied to Scots pine (Xenakis et al., 2008) and Sitka spruce (Minnuno, 2009). The model is based on tree ecophysiology, but with important statistical components also included such as allometric equations, which increase model robustness. The model calculates woody biomass (carbon) outputs at the stand level for even-aged forests and coupled carbon and nitrogen balances in the soil. Thus, 3PGN enables a complete analysis of stand-scale CO_2 fluxes. Outputs allow assessments of short and long term impacts of afforesting peaty soils.

The values for stand productivity and CO_2 fluxes obtained are comparable with those reported elsewhere (Jarvis et al., 2009) for a Sitka spruce stand of moderate productivity (Yield Class 14, i.e. 14 m^3 timber ha^{-1} y^{-1}) on a peaty gley soil producing a net C accumulation during the active growth phase of approximately 27 t CO_2 ha^{-1} y^{-1}. The model can be used to investigate the impacts of forestry on peat soils. In figure 3.1 the changes in net stand and soil CO_2 uptake and total C stocks are compared for Sitka spruce growing on deep peat (top panel YC 10) and peaty gley (bottom panel YC 14).

On the deep peat, the simulation suggests that soil C loss is relatively small (approx. 50 t C ha^{-1}) over the first 5 years, and the total stock change recovers to exceed initial values within 20 years. The time course in the 2^{nd} rotation is very similar. When tree growth on a peaty gley soil is simulated C stocks accumulate in the second rotation with a 7% increase in total C stock despite a small decline in stand CO_2 uptake, thus the net C sequestration benefits are clear over two rotations and are likely to continue, but diminish for deep peats over subsequent rotations.

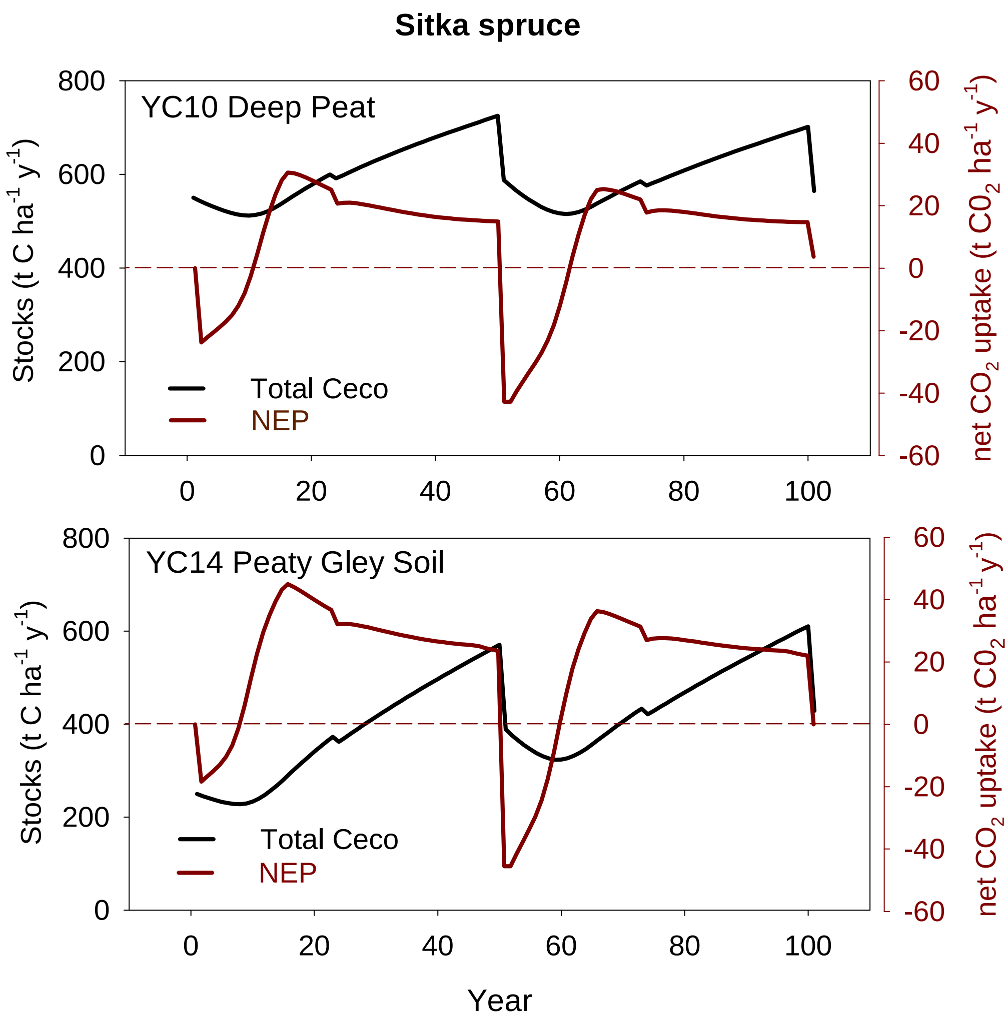


Figure 3.1. Modelled total ecosystem net CO_2 uptake (NEP, t CO_2 ha^{-1} y^{-1}) and total ecosystem C stock changes (Total Ceco, t C ha^{-1} y^{-1}) over two rotations of even-aged Sitka spruce. Soil types were deep peat, [top panel] and peaty gley [bottom panel]. Outputs are from the model 3PGN (after Minnuno, 2009).

4. Estimating forest carbon stocks and changes on peat

4.1 Role of forests in net abatement of GHG emissions

Forestry has substantial potential to contribute to the mitigation of climate change through net abatement of GHG emissions (see Read et al., 2009). There are 3 main contributions forestry can make to abatement: 1) by removal of atmospheric CO_2 during growth of trees; 2) by the provision of low fossil-fuel intensive materials for construction etc., and 3) by the provision of woody biomass for energy generation. However, the abatement potential is reduced by any additional release of CO_2 from SOC and by emissions from use of fossil-fuel in all forestry operations and timber transport (Matthews and Broadmeadow, 2009). Thus estimating net benefits depends on estimating C and GHG balance for the whole sector from tree stand to utilized product. This chapter describes simulations of these GHG balances for stands on peat soils.

4.2 Calculating forest C stocks and changes using C-SORT

The objective was to establish indicative changes in emissions abatement under a number of different management options and woodland creation scenarios. The original remit covered the scenarios of:

- Productive conifer
- Productive broadleaf
- Mixture of conifer/broadleaf
- Native woodland
- Regenerating native woodland

It has not been possible to estimate the latter scenario as the simulation would require new and more detailed information of the woodland characteristics and likely management than is readily available. To address this would require significant additional time and effort and new data collection.

C-SORT Model Outline

The simulations have been undertaken using the Forest Research C-SORT model. C-SORT is the successor to the CARBINE model (Thompson and Matthews, 1989a,b) and has common features of structure and functionality with other forest carbon models such as C-FLOW (Dewar, 1990; Dewar and Cannell, 1992), and CO2fix (Mohren et al., 1999). Links between the forest sector and harvested wood products (HWP) are represented within the C-SORT model. **The model is based on conventional yield models** (e.g. Edwards and Christie, 1981)**, coupled to models of biomass allocation, carbon content, decomposition, soil carbon exchange, product utilisation and empirical data on the GHG balance of forestry operations. It also provides an estimate of the harvested wood products pool, as a guide to cumulative carbon sinks outside the forest**. Since the scenarios are over 200 years, it is unlikely that much timber resulting from the first rotation of a managed forest will exist some 100+ years later, thus we assume a loss rate of timber from the HWP.

The forest biomass and management components incorporated in C-SORT are shown in Figure 4.1. The modeling involves a sequential approach in which an appropriate yield model is selected and used to estimate biomass of various tree components employing the B-SORT model (Matthews and Duckworth, 2005). Wood density (Lavers, 1983) values convert wood volumes to dry weight, 50% of which is assumed to be carbon (Matthews, 1993). The biomass components are roots, stump, roundwood, sawlog, tips, branches and foliage. These components are considered either as standing (living) biomass, in-forest debris, or extracted material to be processed. Alongside growth and product estimations, **the impacts on soil carbon and operational fossil fuel use involved in establishing, maintaining and harvesting trees, (including roads, herbicides etc) are all considered and accounted for.**

Model estimates have been produced for a number of species and site conditions broadly representative of most of the scenarios above. We do not consider the suitability of the species for the sites nor additional factors such as drainage. While site conditions and species are key determinants of C stocks and sequestration potential of woodland, stand management (notably thinning, application of silvicultural systems and rotation length) also has profound impacts. In this series of simulations, the GHG balance of the non-forest understorey is not considered.

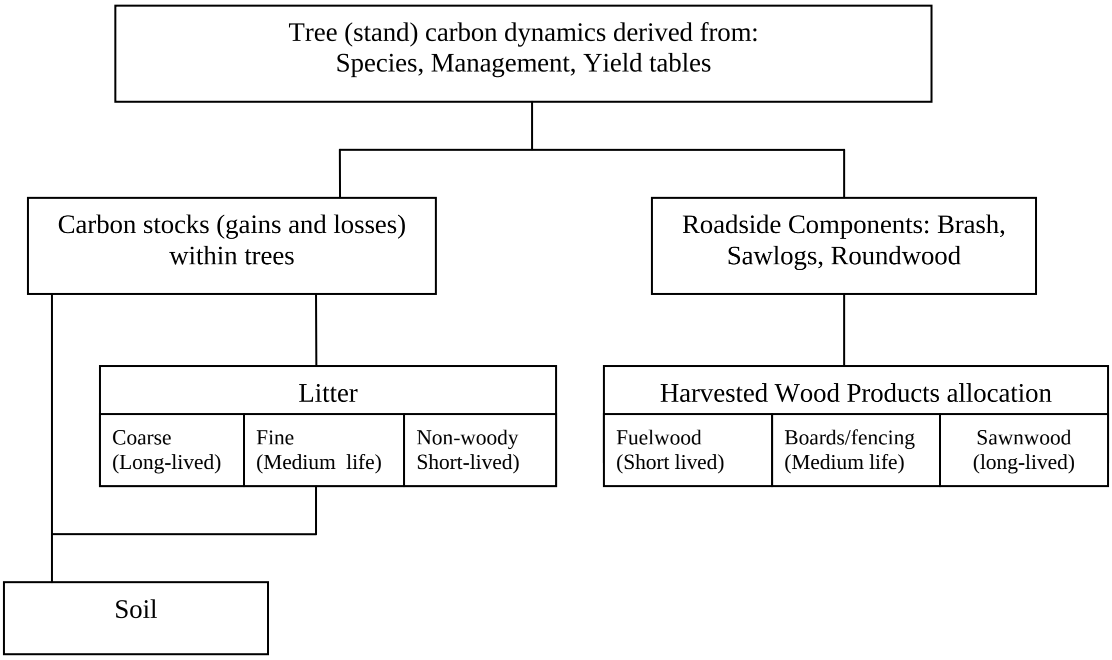


Figure 4.1. Schematic representation of the components within the C-SORT model, with the outputs at the forest road-side.

Basic Model Assumptions

Clearly the range of species, yield class and management types that could be used are very wide for the broad forest type scenarios specified. The scenarios modelled give a wide but realistic cross section of potential management options. In each of the scenarios, the process starts with the selection of appropriate yield table estimates of growth potential. Management is carried out according to normal prescription, including a thinning regime of management table thinning recommendation (Edwards and Christie, 1971). Further details are summarised in Table 4.1.

Table 4.1: Summary of species, growth-rates and the management assumptions used in the C-SORT scenarios shown in Figures 4.2-4.4.

Scenario	Species	Yield Table	Spacing (m)	Management	Soil / previous land use
Productive conifer 1	Sitka Spruce	SS 14	2.0	Standard management. Fell at age 58 y	From pasture
Productive conifer 2	Sitka Spruce	SS 10	2.0	Standard management. Fell at age 62 y	From pasture
Productive Broadleaf	Birch	SAB 4	2.0	Standard management. Fell at age 50 y	From pasture
Mixture	Scots pine (40%) Birch (60%)	SP 4 SAB 4	2.0 2.0	Standard management, synchronise thinning of species. Extend rotation to enable combined fell at Max-MAI of latest species [SP, 95 y]	From pasture
Native woodland	Scots pine (40%) Birch (60%)	SP 4 SAB 4	1.5 1.5	Standard management, synchronise thinning of species. Extend rotation to enable combined fell at age 90 y	From established woodland

Establishment and operational costs

Included with the operational calculations are the fossil fuel combustion (considering all GHG emissions in terms of CO_2 equivalents) associated with road construction and maintenance, ground preparation (including herbicide application), fencing, planting, and beat-up (assuming a 20% loss of plants). Fuel use in ground preparation with ripping and rotovation is assumed to total 0.12 t CO_2eq, equivalent to approx two passes of an agricultural plough (Mason, Nicoll & Perks, 2009, p. 111). Following a clear-fell, a two-year fallow period occurs. Emissions from fossil fuel use in harvesting and extraction to roadside are included. Previous analyses have shown that these **operational emissions are generally small, compared to rates of uptake by productive forests** (see Mason, Nicoll & Perks, 2009 and Section 3.6 previously).

Thinnings and clear-fell

We assume that the first two thinnings are extracted as above–ground, whole trees, and are likely to be used as biomass for woodfuel. Subsequent thinnings and clear-fell are assumed to be a combination of brash material, suitable for fuel, round-wood, and saw-logs. Some material is left in-forest as residue.

Debris

Material from the trees, left in the forest, whether as residue following a thinning/fell or as mortality is allocated to one of three debris pools, each in effect having different rates of turnover, from fast to slow.

Soil

The present soil model in C-SORT uses a quite simplistic approach in line with current paradigm (see Section 2.3) **with three 'pools' of carbon**: an inert portion, a slowly mobilised element, and a highly mobile pool (as described in Section 2.6). However, the parameters defining decays and transfers are empirically derived, and have no environmental drivers or feedbacks. It does not attempt to approach the complexity of well-known but complex and data-hungry soil process models like Roth-C or ECOSSE. Its limitations should be recognised and the outputs should be used as an indicator of the potential scale of soil C stocks and changes (see Chapter 2). **The soil model does not consider effects of water-logging and draining of peaty soils, and does not estimate CH_4 or N_2O emissions**. Soil C stock values used were derived from the BioSoil values presented in Section 2.1, (down to approx. 80 cm) and are appropriate for a peaty gley soil. The same soil was chosen for all the simulations to aid comparison. For the first four simulation conditions, the peaty gley was assumed to be under rough pasture and the C stock in the original vegetation is not included. In the native woodland establishment simulation, it was assumed an existing forest had been clearfelled, resulting in debris addition to the soil, which was close to equilibrium SOC.

Harvested Wood Products (HWP)
The main output of these simulations of extracted woody biomass material is product at forest roadside. At roadside any extracted material is of course un-processed, but it represents a store of carbon until the product decays or is burnt. In order to estimate this C stock, we include HWP as an additional output; its allocation to grouped end-use products is shown in Table 4.2. The HWP profile will be sensitive to end-usage and longevity of the material which will vary for each product. We do not include the operational costs of transportation of material from roadside, nor conversion costs in producing the end-product. Only primary usage is considered; we do not evaluate the potential of re-cycling material.

Table 4.2: Assumed allocation of extracted roadside material to harvested wood products (HWP)

Extracted Material	**Softwood HWP**			**Hardwood HWP**		
	Timber	Board/fencing	Fuel	Timber	Board/fencing	Fuel
Sawlogs	45%	18%	27%	55%	25%	20%
Roundwood		70%	30%			83%
Brash etc.			100%			100%

Product life-span:
Fuel: all material is assumed to be consumed within a year.
Board/Fencing: we assume a 30 year period before any decomposition, followed by an exponential reduction, such that 95% of material has been lost 50 years after harvest.
Timber products: we assume a 60 year period before any decomposition, followed by an exponential decay, such that 95% of material has been released over the next 30 years.

4.3 C-SORT forest C stock results

Figures 4.2-4.4 show results from the simulations described in Table 4.2. **It is clear that the large SOC stock of the peaty gley soil assumed for these simulations** (310-360 t C ha^{-1}, see Section 2.2) **dominates the total C stock in the above ground forest, soil, and HWP throughout the time courses, and for all simulations.** For the highest-yielding Sitka spruce simulation (YC 14, Figure 4.2 upper) the above-ground biomass (with associated debris) reaches a maximum of about 150 t C ha^{-1}. This results in a peak forest C stock of 550 t C ha^{-1}. (Note: this is similar to the C stock simulated in the very different model 3PGN, shown in Fig 3.1). Including the HWP ('product') increases this up to a maximum of approximately 600 t C ha^{-1}, thus making the above-ground component a maximum of 25% of the total C stock. Calculated C accumulation in the above-ground biomass of Sitka spruce YC 10 (Figure 4.2 lower) is about 20% lower than YC 14 at peak and the rotations are longer. Peak above-ground biomass in the birch simulation (YC 4, Figure 4.3 upper) is substantially smaller, and there is little HWP compared to the Sitka spruce simulations, and shorter rotations. The mixed Scots pine and birch simulation (Figure 4.3 lower) has a long rotation and thus a long period with substantial standing biomass compared with the birch scenario. The native woodland scenario (Figure 4.4) also produces longer-standing biomass, the key difference from the mixed species scenario being the pool of HWPs available at the beginning from the previous clearfell, and that the soil C is already assumed close to equilibrium with woodland land use. **In all the afforestation-from-grassland scenarios the simulation shows that soil C increases, the rate of increase being linked to the productivity**. In the native woodland scenario which is a restock, soil C is simulated to drop slightly, before recovering. The right hand sets of figures show the total and in-forest values with the emissions from forestry operations included (see above explanation). However, in all cases these emissions are small and apparently make little difference to the time-courses.

Comparing outputs from these simulations can be challenging as the time-courses are different, so that comparison at any one time will include a different number of complete and partial rotations. Table 4.3 shows one set of summaries of the simulations. The Sitka spruce YC 14 has a total of 779 t C ha^{-1} extracted, compared to 365 t C ha^{-1} from a sycamore/ash/birch YC 4 forest. The model predicts similar increases in soil C in these two cases, because of the accumulation of organic matter as litter and modelled incorporation into SOC. Where productivity is lower (scenario for SP/SAB mix the C losses from establishment and management operations relate closely to the amount of material extracted, as expected. However, when the total C uptakes and losses, including operational costs are summed over the simulation period of 200 year, the average rate of increase in C stock is very similar. However, such average values will be sensitive to the assumptions of decay rates of the extracted products. Since the yield from the spruce is higher, then it is evident that the pool of existing product (HWP) will be higher, although by the end of the simulated period, most of the material from the first rotation will no longer be included in the C stock.

Table 4.3: Summary of simulated carbon stocks over a 200 year period for 5 different woodland creation scenarios.

	Soil C change t C ha^{-1}	**Average Standing stock** t C ha^{-1}	**Average Debris stock** t C ha^{-1}	**In-forest Operations emissions** t C ha^{-1}	**Average C increase** t C ha^{-1} y^{-1}	**Total Extracted** t C ha^{-1}
SS 14	+82	53	10	-20	0.62	779
SS 10	+60	45	8	-17	0.48	595
SAB 4	+84	35	10	-11	0.59	365
SP/SAB	+58	46	7	-10	0.50	310
Native mix SP/SAB	+7	46	8	-10	0.21	312

If the consideration was only to be based upon maximising C stock, these calculations suggest that if a high yielding spruce could be established then it would produce the highest return in C accumulation. However, if the site is unlikely to produce such a high yielding crop, then it may be better for a broadleaf such as birch crop to be established. This takes no account of financial implications of managing and selling the crop, nor considering if occurrences such as wind-blow bring an early conclusion to the rotation.

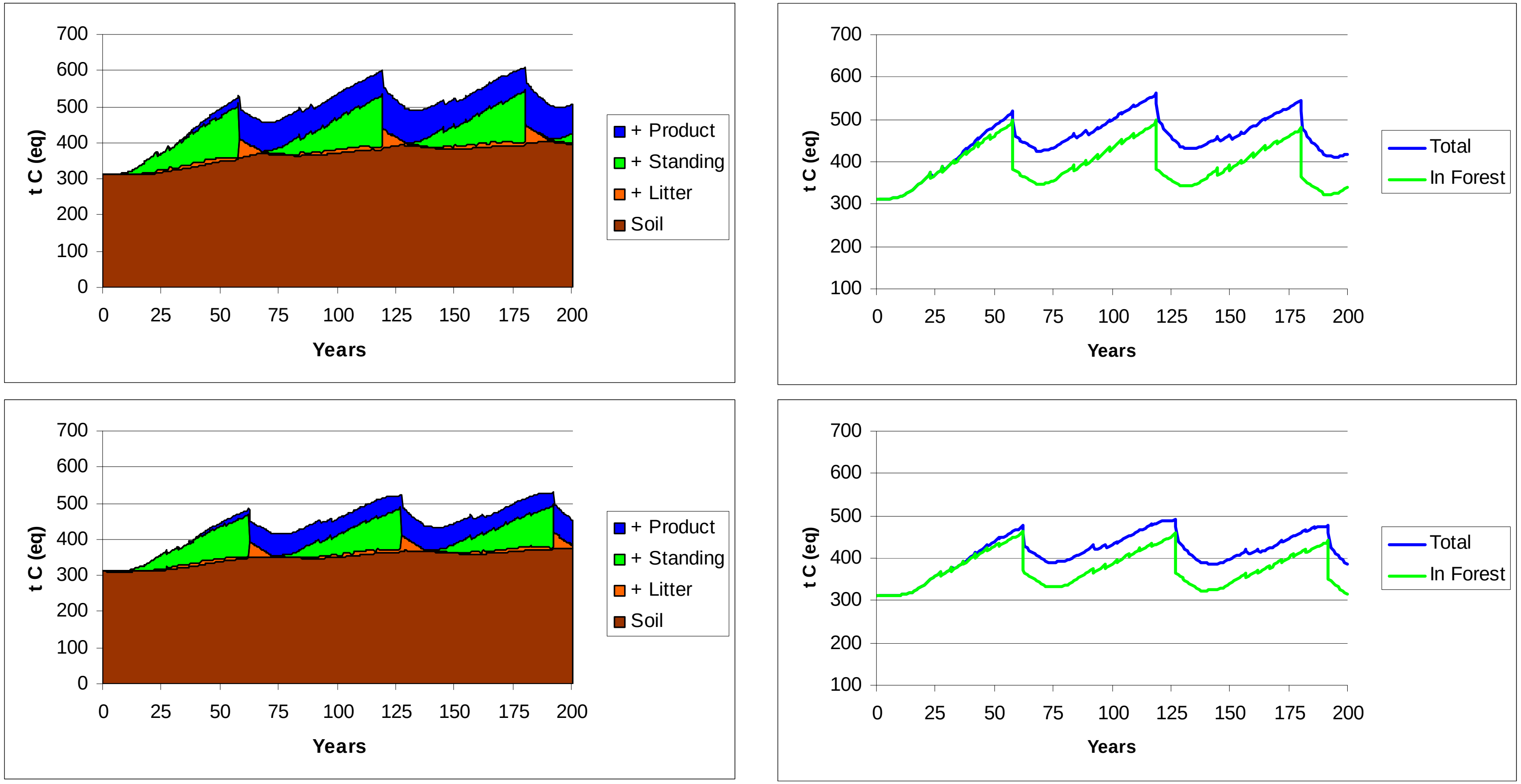
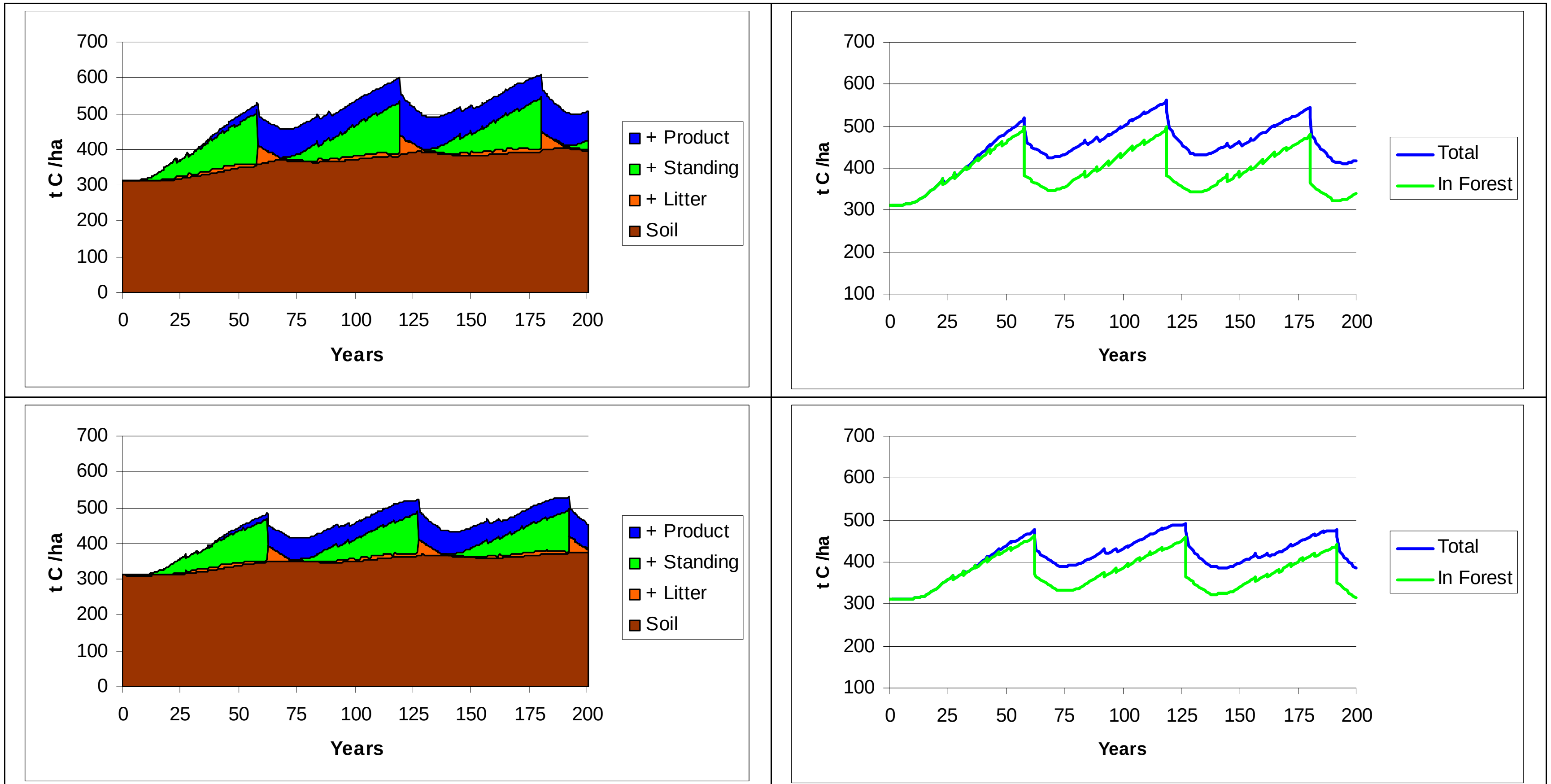


Figure 4.2: Simulations from C-SORT for establishing a Sitka spruce forest on a peaty gley soil, originally under rough pasture, over a 200 year period. Top figures: Yield Class 14; Lower: Yield Class 10. LH figures show the cumulative C stocks (t C ha^{-1}) including different components. RH figures are the same values but adjusted for emissions from forestry operations.



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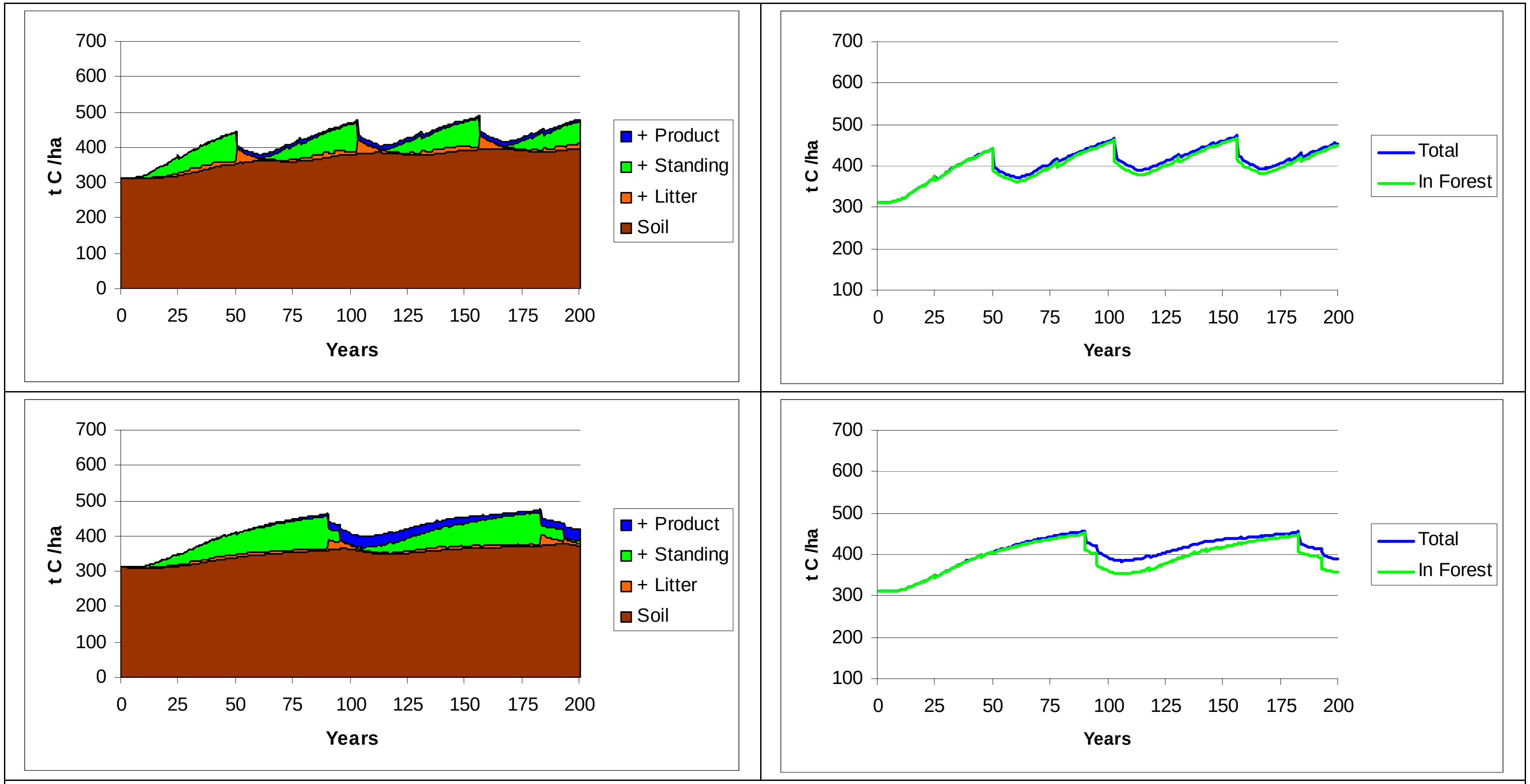

Figure 4.3: Simulations from C-SORT for establishing woodland on a peaty gley soil, originally under pasture, over a 200 year period. Upper figures: birch, Yield Class 4. Lower figures: a Birch/Scots pine mixture both Yield Class 4. LH figures show the cumulative C stocks including different components. RH figures are the same values but adjusted for emissions from forestry operations.

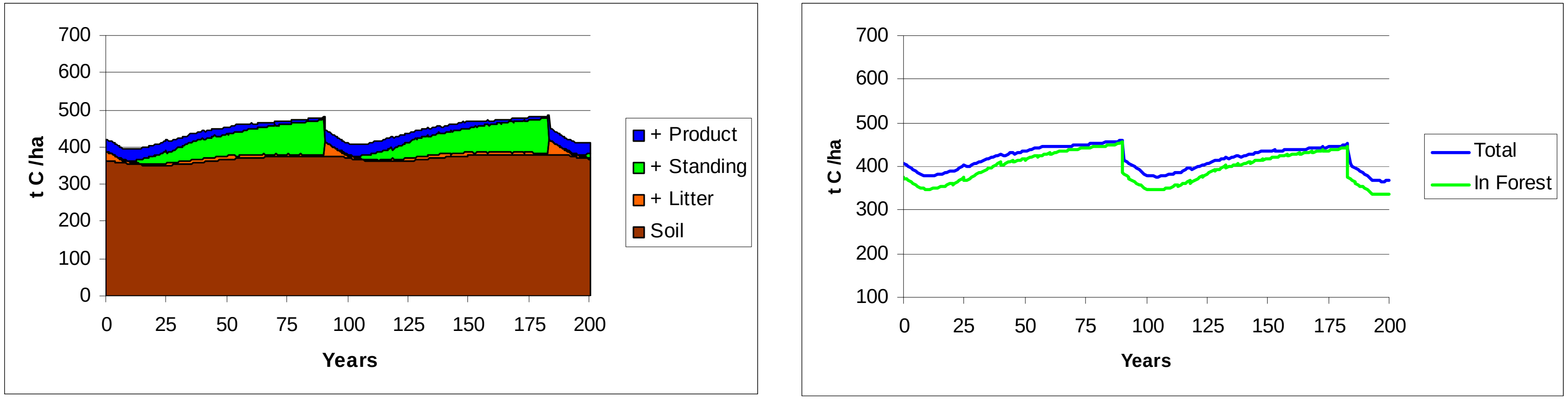

Figure 4.4: Simulations from C-SORT for establishing native woodland on a peaty gley soil with a birch and Scots pine mix over 200 years, assuming existing forest. LH figures show the cumulative C stocks including different components. RH figures are the same values but adjusted for emissions from forestry operations.

5. Summary & Conclusions

It is well documented and understood that peat soils contain large stocks of organic carbon (SOC), which are altered by changes in land use, including forestry. Soil organic carbon can be lost by decomposition and gaseous CO_2 efflux (Section 3.1), particulate erosion (POC) or dissolved in rainwater drainage and runoff (DOC, Section 2.6). Conversely, forest growth can lead to an accumulation of soil C, through litter formation and incorporation. Understanding the consequences of forestry activity on peat soil C stocks and GHG balances depends upon four important aspects:

- the type of peat soil and proportions of different SOC fractions,
- whether previously planted and how prepared or cultivated,
- the level of disturbance during planting,
- and the modification to the water table depth.

It is very difficult to quantify the extent to which tree planting on peat soils results in loss of SOC and changes to GHG balances as there are very few definitive data (Chapter 3). It would be unwise to rely on the few studies that exist in UK conditions to make a general statement that would be appropriate for all time-frames. There is similarly rather little and somewhat conflicting appropriate information on GHG emissions from peatlands, in part because of the variety of sites and conditions investigated. In particular, there is little conclusive information on GHG fluxes from completely pristine peatbogs in the UK. It is therefore not possible to state with confidence whether the net GHG emissions from afforested peatland in UK conditions, over the complete forest planting-harvesting cycle are less than those from pristine peatlands, or other common peatland land uses.

However, it is very probable that moderate and good productivity forests planted on shallower peat soils with limited disturbance provide a net C uptake over the cycle, and because of the reduction in CH_4 emissions with afforestation, therefore are a net sink for GHG. On deeper peat soils the possibility of continuing losses of SOC combined with usually lower tree growth rates, means that net GHG emissions abatement possibilities are likely to be smaller, and in some cases could be negative.

Understanding the net C and GHG balance of a forest stand and the soil requires use of detailed process models (Section 3.7), and these need to be sufficiently comprehensive to include CH_4 and N_2O fluxes and be appropriate for organic soils in UK conditions. Appropriate models are only now becoming available. In addition, the overall GHG balance benefits of CO_2 uptake by the forest and production of timber and/or woodfuel should be considered (Section 4). This requires linking existing robust forest C and GHG accounting models with appropriate soil models.

The main points emerging from the analysis presented here are:

Policy and Operationally Relevant points

1. Average soil organic C stocks under Scottish forest stands to 1 m on *shallow peat* soils (those with <40 cm depth of peat layer) are 350 t C ha^{-1}, *deep peat soils* (those with >40 cm depth of peat layer) contain 510 t C ha^{-1} (Section 2.2).
2. Total areas of high SOC soils, in Scotland, are 30,094 km^2 of shallow peat soils and 8,818 km^2 of deep peat soils, with currently 20% and 17% of these areas are under woodland, respectively (Table 2.1, Section 2.3).
3. Deep peat areas are predominately in Dumfries & Galloway, Grampian Highlands and NE areas of Scotland (Figure 2.3, Section 2.4).

4. It is estimated that deep peat soils under forest in Scotland contain 76 Mt C (down to 1 m) and shallow peat soils 207 Mt C. These are 7% and 19% of the total C stocks in peat soils in Scotland. Below 1 m, deep peat soils may contain another 35 Mt C (Section 2.4).
5. Restricting planting of new woodland in Scotland so that deep peat soils were not used would remove between 7,400 and 10, 300 km^2 of presently unplanted deep peat from potential afforestation, affecting a potential SOC stock of 702 Mt C (Section 2.4).
6. For current forested areas in Scotland (conifer and broadleaved) not restocking deep peats at the end of the rotation would affect 11% of the Scottish forested area, but this area comprises 27% of the forested peat soil C stock (Section 2.4).
7. Projected climate change for Scotland over the next 50 years are likely to cause small reductions in overall peat SOC stocks, although there is the possibility of larger losses due to droughts which, if prolonged, may cause permanent changes to peatlands and due to local wildfires (Section 3.3).
8. The GHG balances of peat soils are affected by afforestation and replanting. Drainage is likely to lower the water table, replacing net CH_4 emissions with net CH_4 uptake, but increasing substantially peat decomposition, CO_2 emissions and DOC losses, as well as affecting N balances (Sections 3.1 & 2). There are tree survival and growth benefits from some form of cultivation in all but the most fertile of sites (Section 3.4), consequently most planting will result in some soil disturbance and loss of SOC.
9. Excavator mounding ground preparation is the dominant cultivation for planting on peaty soils which improves establishment conditions; this causes less disturbance than ploughing used previously. However, on a deep peat soil, mounding (particularly if combined with trenching) will cause a substantial local water table drop which will reduce SOC stock (Section 3.5).
10. For new woodland establishment such standard mechanical ground preparation and efforts to improve drainage are likely to result in high SOC losses on all deep peats. The net GHG balance of new planting on very wet and poor deep peats is likely to be particularly poor, due to the high likelihood of soil C loss, low expected tree yield and likely need for fertilisation (Sections 3.5 & 6). However, in some cases there might be net GHG and SOC balance benefits in afforestation of degraded flushed blanket bogs, such as those that have either been modified by agriculture or previous peat extraction (Section 3.5).
11. The net GHG balance benefits of restocking previously planted and disturbed peat soils are very different from those of new planting on peat soils which currently support non-woodland vegetation (Section 3.5). If tree growth is good, and establishment can be successful with little disturbance, there is some evidence that restocking is likely to recover the SOC losses during the first rotation (Sections 3.6) which is supported by preliminary modelling combining stand and soil dynamics (Section 3.7).
12. However, on restock sites excavator mounding may bury surface OM and prolong residence times. On shallower peaty gley soils, mounding may also mix peat with mineral layers, and the effect on decomposition, SOC stability and soil C loss is hard to predict (Section 3.6).
13. Restocking of wet and low fertility sites, with low forest yield potentials is not likely to increase SOC stocks whereas restocking of sites with higher yield potentials, and favourable soil types with minimum fertilisation needs, may produce continued overall GHG balance benefits from a second rotation (Sections 3.5 & 6).
14. The overall GHG balance benefits of CO_2 uptake by the forest and production of timber and/or biomass should be considered (Section 4); reliable models are required for this, but as yet forest C accounting models do not include detailed soil C changes and complete soil GHG balance calculations. Simulations of the overall C balance for woodland creation on a peaty gley soil with the C-SORT model emphasise that above

ground stocks for yield class 14 Sitka spruce are at maximum only 25% of the total C stock in soils, trees and HWP. For lower productivity woodland scenarios the calculations show that C stocks and emissions abatement benefits are substantially smaller (Section 4.3).

Additional Scientific Points:

15. The accuracy of up-scaling of SOC stocks depends critically on the level of soil mapping detail and the accurate determination of peat depth and bulk density profiles (Sections 2.4 and 2.5).
16. Assessing SOC totals alone are not sufficient to understand changes in C stocks, as SOC is held in different fractions, with very different characteristics and residence times affecting their stability. SOC changes depend also on soil N content, litter input and its quality and quantities, and environmental conditions – particularly precipitation and temperature.
17. The fast-turnover free particulate organic matter (FPOM) fraction is large in organic soils, while the slower turnover occluded particulate and mineral associated OM fractions (OPOM and MAOM) are higher in mineral soil horizons (Section 2.6)
18. Due to the soil organic matter origin, fate and composition, forest floors and organic soil horizons are more likely to lose carbon than underlying mineral soil layers when disturbed (Section 2.6). Mineral layers may retain dissolved organic carbon lost from higher layers.
19. Dissolved organic carbon (DOC) fluxes from organic soils are likely to increase on disturbance, but are usually small compared to other C balance components (Section 2.6).
20. Soil N content is strongly linked to C content, so C amount is important in determining N loss or retention, and *vice versa*. High N deposition from pollution may enhance peat decomposition and SOC loss (Section 2.6).
21. The N_2O fluxes from peatlands are usually a small component of the total GHG balance; CH_4 can be much more important (Section 3.2). Peat soils typically emit CH_4 when wet, and the emission rate depends critically on water table depth. The top 0-20 cm layer dominates the soil CO_2, CH_4 and N_2O emissions and consumption, although for CH_4 production can occur much deeper. Therefore the depth of peat beyond this layer has little effect (Section 3.1).

Implications for the FCS Interim Policy Statement on peat soils

This report shows that despite the lack of comprehensive and unequivocal quantitative evidence on the impacts of afforestation on peat soil C stocks and GHG balances, there is sufficient overall understanding of the subject to inform policy recommendations. The key points are:

- Enhanced soil C loss will occur when peaty soils are disturbed.
- As forestry activities on peat will result in some disturbance there will be increased soil C loss, which will significantly reduce the net C uptake of the site and the wider net GHG emissions abatement benefits of forestry.
- Thus minimising disturbance is essential.
- On deeper peats with low tree productivity and possibly larger C losses continuing over long periods the net C uptake may be small or even negative, depending on the duration of tree growth. This means that low or even negative net C accumulation is likely, and the net GHG balance may be poor, even taking into account the likely reduction in CH_4 fluxes under forest.
- Therefore, in view of the large amount of soil C presently stored in deep peats in Scotland (relative to the land area involved), restricting new planting to shallower peats (<50 cm deep) with less potential C loss, and usually better tree growth conditions is a sensible precaution.

- Consideration should be given to aligning the depth limit in the Policy Guidance with that used in the FC soil classification for deep peat: > 45 cm organic matter.

6. Research Recommendations

The above sections have highlighted that there are several key gaps in knowledge that prevent definitive statements on the risks of disturbance to peat soils of particularly depths through forestry (or other land uses). These gaps are:

1. Determination of the different OM fractions in peaty forest soils under different forest species and peat types, which are required to model soil C dynamics reliably
2. Quantification of the stability and decomposition rates of peat OM from forest and other vegetation inputs and their components and fractions in appropriate environmental conditions.
3. Determination of effects of afforestation on peaty soil C stocks for a much more complete range of soil, forest types and ages. Because of the need to assess relatively slow changes, ideally, paired sites approaches and carefully controlled chronosequences should be used. However, these will be necessarily limited to information about *previous* afforestation practices.
4. Much more complete evidence on the effects of typical *current* soil preparation practice on soil C stocks and losses, and on soil GHG fluxes.
5. More evidence on the effects of typical thinning and clearfell practice on soil C stocks and losses, and on soil GHG fluxes.
6. Reliable models for SOC turnover and changes and consequent C losses and GHG fluxes during land-use changes such as afforestation on peat soils. Recent models have now started to attempt to incorporate quantification of the various soil modification processes that occur when e.g. grassland is converted to forest, but the research is only starting.
7. Comprehensive simultaneous determination of GHG fluxes (CO_2, CH_4 and N_2O) from peat soils under forest and other land uses, at different times during the stand rotation. Clearly, it is unrealistic to hope to cover experimentally the different range of land use, forest management, seasonal and environmental conditions. Therefore it will be necessary to use advanced process-based models (e.g. DNDC and/or ECOSSE) that can simulate GHG fluxes and soil and vegetation C stock changes reliably to do fuller assessment of risks to organic soil C stocks, and net GHG balances. Ideally, in order to derive sufficient information to calibrate these models, example consistent data sets of GHG fluxes in forests and other land uses are required in a range of appropriate climate and conditions. When calibrated and evaluated, such models can then also be used to assess impacts of projected climate change.
8. Detailed soil C & GHG balance models need to be coupled to forest C accounting type models in order to explore quantitatively the effect of afforestation on net GHG balances, and GHG emissions abatement potential.
9. Forest process-based and empirical C accounting models have been largely derived and tested in the UK on managed woodland, and in particular on monospecific, even aged commercial stands. Basic information on the growth and C uptake characteristics of semi-natural and native species woodlands is required in order to be able to produce reliable model estimates for these types, particularly where understorey may be an important component.

It should be pointed out that some of the necessary research is already underway, either within forestry-specific research, or other peatland, land-use, soils or environmental research. A wide assessment of relevant work planned or underway would be a useful exercise, although it was outside the scope and resources of this project. Some of the gaps could be filled by the research identified relatively quickly (short-term, 6-18 months, nos. 1-4 above); others would ideally require longer-term research projects lasting from 18 months to several years (nos. 5-9 above).

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Glossary

FPOC	Free particulate organic carbon
GHG	Greenhouse Gas
GWP	Global Warming Potential, defined as the contribution to cumulative warming over time, usually 100 years, for a particular GHG, relative to CO_2. According to the IPCC (2007) the GWP is equal to 1, 25 and 298 for CO_2, CH_4 and N_2O respectively.
MAI	Mean Annual Increment
MAOM	Mineral absorbed organic matter
Mineratrophic	ecosystems deriving all water and nutrients from groundwater or streams, e.g. fens.
NEP	Net Ecosystem Production, a measure of the net uptake of CO_2 into vegetation and the underlying soil.
OM	Organic Matter/Material
Ombrotrophic	ecosystems deriving all water nutrients from precipitation – e.g. much blanket bog.
POC	Particulate Organic Carbon
Screefing	removal of herbaceous vegetation and soil organic matter to expose a soil surface for planting.
SOC	Soil Organic Carbon
SOM	Soil Organic Matter/material (i.e. not just C component)

Appendix A

Draft of 22 January 2010.

FCS interim policy statement on woodland creation on deep peats

Until more definitive evidence is established on the net greenhouse gas implications of ground disturbance and tree planting on organic soils we will operate an interim, general presumption against new woodland creation on soils with peat exceeding 50 cms in depth.

Exceptions may be appropriate, at the discretion of Conservators, for limited areas of woodland expansion which are likely to have both significant/high environmental or scenic benefits and relatively low potential impacts on greenhouse gases in terms of ground preparation intensity, fertilising and/or percentage canopy cover. The most likely cases to meet these criteria are new native woodlands, established both through planting and/or natural colonisation, especially where the areas of deep peat (> 50cm depth) are an integral part of a scheme which is predominantly on soils with shallower peat or mineral soils, or where the intention is to establish missing components of priority woodland such as montane scrub or treeline woodland.

As is already the case, the biodiversity and cultural heritage values of some sites might mean that new planting may still be constrained on some shallower peats sites.

Policy on re-stocking deep peats is set out in the Scottish Government's policy on control of woodland removal.

Appendix B

Greenhouse gas fluxes reported for UK forest sites and other vegetation, particularly on peaty soils. Negative values indicate uptake by the soil, positive values indicate emissions. (from Jarvis et al., 2009 and Morison et al., 2010).

Activity / Site	soil type	Species	N_2O (kg ha^{-1} y^{-1})	CH_4 (kg ha^{-1} y^{-1})	CO_2 (t ha^{-1} y^{-1})	CO_2eq (t ha^{-1} y^{-1})	References
Standing forest							
Glencorse	silty loam, brown forest soil		0.12–0.28				Kesik et al., 2005
Glencorse Dunslair Heights Central Scotland, Devilla North Berwick Dunslair Heights Auchencorth Moss North England; Great Dunfell Dunslair Heights Auchencorth Moss	silty loam, brown forest soil peaty podzol drained peat peat peaty podzol peat	 Sitka spruce alder birch Sitka spruce pine sycamore Sitka spruce beech upland grass grass/heather grass/sphagnum	 0.55 2.07 0.94 0.2 0.57 1.04 0.49 0.36 0.83 0.41 0.16				Skiba et al., 1996
Ireland	low humic mineral gley in 15 and 13 yr old stands	Sitka spruce 10, 15 31 & 47 yr old stands - undisturbed ground - ridges - furrow			 25.7-48.5 21.3-340.8 35.2-36.9		Saiz et al., 2006a,b
Cloosh Forest, Co. Galway, Ireland	ombrotrophic blanket peatland 0.9 to 5.5 m depth	- recently planted Sitka spruce - lodgepole pine 19-33 yr old - mature Sitka spruce			6.2 3.7-5.1 9.5		and Farrell, 2005

Byrne

Harwood 20yr-30yr and yr1-yr2	fine loam over clay with peat surface horizon 35-50-cm		0.2–4.7	0.2–2.8	7.8-22.3	7.9-23.8	Ball et al., 2007
Gisburn Forest; clear-felled and replanted in 1991	cambic stagnogleys to stagnohumic gleys	no statistically significant differences between species (alder, oak, spruce and pine)		-0.51 Mean forest			McNamara et al., 2008
Harwood forest, Northumberland	organic rich peaty gley (Histic Gleysols)	40 yr Sitka spruce	1.01	–1.6	17.5	17.8	Zerva and Mencuccini, 2005
Kershope	peaty gley		2.0–4.1				Dutch and Ineson, 1990
7 UK forest and woodland sites	various - unstated			–0.1 to –9.1 Median: –2.4			Smith et al., 2000
Clearfelled sites (CF)							
Harwood yr1-yr2	fine loam over clay with peat surface horizon 35-50-cm		0.7–0.9	6.8–18	23.7-26.0	24.1-27.1	Ball et al., 2007
Kershope	peaty gley		12––51				Dutch and Ineson, 1990
Cloosh Forest, Co. Galway, Ireland	ombrotrophic blanket peatland 0.9 to 5.5 m depth				5.1-5.9		Byrne and Farrell, 2005
Harwood	organic rich peaty gley (Histic Gleysols)		2.5	8.8	11.83	12.8	Zerva and Mencuccini, 2005
Other vegetation sites							
Harwood yr1-yr2	fine loam over clay with peat surface horizon 35-50-cm	Unplanted grassland	0.3	1.2-2.6	33.1-55.8	33.2-56.0	Ball et al., 2007

Auchencorth Moss	Acid peatland, 85% histosols "peats"; 9% Glaysol; 3% Humic Gleysol, 3% Cambisol; peat depth <0.5 to >5m with low-intensity sheep grazing and peat extraction		mean ±SE				
		Calluna	0.13 ± 0.29	0.7 ± 0.5	25.4 ± 3.5	25.5	Dinsmore et al., 2009
		Hollow	-0.10 ± 0.13	1.8 ± 2.1	21.0 ± 0.9	21.0	
		Sedge/Hummock	0.18 ± 0.17	0.2± 0.6	21.0 ± 1.8	21.1	
		Juncus/Hummock	-0.06 ± 0.12	0.4 ± 0.6	23.4 ± 9.6	23.4	
		Riparian	0.34 ± 0.12	51.3 ± 27.2	39.4 ± 5.3	40.8	
UK	nutrient poor bogs	unmanaged wetland		13.3-53.3			Joosten and Clark, 2002
southeast Scotland	montaine soil very peaty with enhanced N deposition of 24.3 kg N ha^{-1} yr-1	moorland vegetation (calluna sp., grasses, mosses Sitka spruce	1.86 0.8				Skiba et al., 1994

Appendix C

The FC soils classification system for the main mineral, shallow peat and deep peat soils (from Kennedy, 2002).

Table 1. The FC classification system for the main mineral & shallow peaty soils

	Soil group	Soil type	Code
Soils with well aerated subsoil	1. Brown earths	Typical brown earth Basic brown earth Upland brown earth Podzolic brown earth	1 1d 1u 1z
	3. Podzols	Typical podzol Hardpan podzol	3 3m
	4. Ironpan soils	Typical ironpan soil Podzolic ironpan soil Intergrade ironpan soil	4 4z 4b
	12. Calcareous soils	Rendzina Calcareous brown earth Argillic brown earth	12a 12b 12t
Soils with poorly aerated subsoil / Gleys	5. Ground-water gley soils	Typical ground-water gley	5
	6. Peaty (surface-water) gley soils	Typical peaty surface-water gley Podzolic peaty surface-water gley	6 6z
	7. Surface-water gley soils	Typical surface-water gley Podzolic surface-water gley Brown surface-water gley	7 7z 7b

Table 2. The FC classification system for deep peats

	Soil group	Soil type	Code
Flushed peatland	8. *Juncus* (or basin) bogs	*Phragmites* (or Fen) bog *Juncus articulatus* or *acutiflorus* bog *Juncus effusus* bog *Carex* bog	8a 8b 8c 8d
	9. *Molinia* (or flushed blanket) bogs	*Molinia, Myrica, Salix* bog Tussocky *Molinia* bog; *Molinia, Calluna* bog Tussocky *Molinia, Eriophorum* bog Non-tussocky *Molinia, Eriophorum, Trichophorum* bog *Trichophorum, Calluna, Eriophorum, Molinia* bog (weakly flushed)	9a 9b 9c 9d 9e
Unflushed peatlands	10. *Sphagnum* (or flat or raised) bogs	Lowland *Sphagnum* bog Upland *Sphagnum* bog	10a 10b
	11. *Calluna, Eriophorum, Trichophorum* (or unflushed blanket) bogs	*Calluna* blanket bog *Calluna, Eriophorum* blanket bog *Trichophorum, Calluna* blanket bog *Eriophorum* blanket bog	11a 11b 11c 11d
	14. Eroded bogs	Shallow hagged eroded bog Deeply hagged eroded bog Pooled eroded bog	14 14h 14w