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Afforestation and restocking on peaty soils – new evidence assessment

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Summary

This report examines new evidence published since the previous Forest Research report 'Understanding the GHG implications of forestry on peat soils in Scotland' (Morison *et al.*, 2010, hereafter referred to as 'FR 2010'). In the interim, new experimental studies, soil resurveys and model application have been commissioned by Forestry Commission Scotland and England, some new literature is available, and guidance on restocking of deep peats has been developed (Forestry Commission Scotland, 2015).

1.1 Key findings

- This review broadly confirms the findings of the 2010 report. It remains 'very probable' that moderate and high productivity forests planted on shallower peat soils with limited disturbance provide a substantial net carbon uptake over the forest cycle. This is because uptake of carbon dioxide by the forest, and its subsequent transfer into the soil, is greater than losses from soil decomposition.
- The net uptake of carbon by the second rotation of forestry on shallow peat soils will balance out the losses of carbon from soil during the first rotation. New evidence since FR 2010 confirms that there is likely to be a substantial reduction in methane emissions with afforestation (if drainage sufficiently lowers the water table), which is an additional GHG balance benefit.
- Overall, site carbon stocks are positive for shallow peat soils over multiple rotations of forestry. The carbon lost through leaching, oxidation and decomposition from peat layers due to the disturbance by soil preparation for afforestation, clear-felling and reforestation could be compensated for by the carbon accumulation in upper organic soil horizons
- For organic (deep peat) soils little new evidence is available, and the majority of studies still show that afforested drained peats dependent on the forest growth and yield class are likely to act as net carbon sinks despite large peat losses. More studies and data on forestry on deep peat are needed.
- Recent reports confirm that ground preparation techniques for tree planting, such as drainage and ploughing increase the overall net release of greenhouse gases when compared with

undrained, undisturbed peaty soils – more carbon dioxide, nitrous oxide and dissolved organic carbon (DOC) loss, but less methane. Work since 2010 strengthens the evidence of a threshold water table depth for methane production over methane removal.

- We found no new literature on thinning effects on shallow or deep peat soils. Clearfelling was found to cause loss of soil carbon stocks on shallow peats due to soil disturbance. However, this loss was compensated through the second rotation forestry.
- The impact of removal of brash (branches and tree tops left after harvest) on soil carbon stocks depends on soil type. New evidence suggests that brash removal on shallow peat results in higher soil C stocks than retention as there is less subsequent decomposition of the peat layer.
- New evidence suggests that stump removal results in high loss of carbon from both shallow and deep peat soils.
- Development of a new Forest Research soil carbon dynamic model has been used for the LULUCF (Land Use, Land Use Change and Forestry) reporting and is coupled for the first time with the forestry productivity model, capturing the changing growth of forest and carbon inputs to soils across forest rotations.

We found new evidence to support:

- Revised and increased estimates of native broadleaved woodland extents, due to new and higher resolution national forest inventory (NFI) work and peat-edge woodland recording, providing a more accurate picture of current forestry on shallow and deep peats.
- Revised and increased estimates of soil carbon stocks, resulting from increased sampling of forested shallow and deep peat soils.
- A better understanding of soil carbon dynamics and the impacts on carbon stocks in second rotation forest. This suggests that a second rotation could balance out the initial loss of shallow peat soil carbon stocks in the first rotation and supports previous research findings.

1.2 Evidence gaps identified

- There is a clear need for long term studies using different planting ages (chronosequence studies) to ensure robust results when evaluating the impacts of afforestation and restocking on soil carbon stocks, as short-term impact studies are likely to provide misleading conclusions.
- There is a need for more quantitative evidence on full greenhouse gas budgets (carbon dioxide, methane and nitrous oxide), microbial process and emission pathways on planted peaty soils in order to assess the afforestation impacts and aid model development and verification.
- Assessing soil greenhouse gas balance requires models which account for methane and nitrous oxide in addition to carbon dioxide.
- A future challenge identified for forest soil-carbon modelling, especially on shallow peat soils, is to include the different stabilisation mechanisms of organic carbon in both the peat and the mineral layers.

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2 Introduction

2.1 Importance of peatlands and their greenhouse gas (GHG) exchange balance in Scotland

The Scottish Forestry Strategy[1] (2006) recognises the wider social and economic benefits that new forestry can deliver and states the Scottish Government's ambition for an increase in woodland cover in Scotland to 25% by the second half of the century. The Scottish Climate Change Plan (2017) has an ambition to increase from the current 18% to around 21% of woodland cover by 2032. This aspiration aims to increase new woodland creation to 15,000 ha of new woodland per year from 2024-25. It is estimated that this will lead to an equivalent of an additional 0.7 Mt y^{-1} of carbon dioxide (CO_2) sequestration by 2020 and an additional 4.4 Mt y^{-1} by 2050 over and above the sequestration by existing woodlands. This will make an important contribution to reducing Scotland's net greenhouse gas (GHG) emissions.

Given that approximately two-thirds of Scotland is covered by high carbon-content organic soils of varying depths, including nearly a quarter with deep peat soils (Chapman et al., 2009), it is important to understand the consequences of forestry activity, which depends upon four important aspects:

1. the type of peat soil and proportions of different organic carbon fractions, whether previously planted and how it was prepared or cultivated,
2. the level of disturbance during planting and subsequent management (thinning/felling), and the modification to the water table depth,
3. the rate of CO_2 uptake by the trees and soil litter through photosynthesis, growth and senescence,
4. the accumulation or use of harvested wood products and their possible net GHG emission reduction benefits through substitution for fossil fuel intensive materials and energy provision.

Afforestation of any organic soil requires extensive land preparation in order to obtain the desired environment for tree establishment and good growth, including the lowering of the water table by drainage. This disturbance can lead to increased carbon (C) loss to the atmosphere as well as to streams and groundwater, but may also result in an enhanced carbon sink in plant biomass, which may lead to more carbon input to the soil.

It is very difficult to quantify the extent to which tree planting on peat soils results in a loss of soil C and changes to GHG balances in Scotland or in other areas of the British Isles or elsewhere with similar conditions. The available data suggest that in afforested soils the loss of CO_2 from the soil dominates in the total contribution to the GHG balance, although methane emissions can be substantial in the wettest sites (FR 2010).

Afforestation on deep peat soils could cause significant losses of soil C through stimulating C loss due to drainage and soil disturbance at planting, whilst planting on shallow peat soils is less likely to lead to significant losses (FR 2010). The carbon changes following a second rotation on deep peat soils are less well understood. However, guidance has been developed to underpin management decisions about restocking on deep peat soils in Scotland (FR 2010; Forestry Commission Scotland, 2015). This is now leading to consideration of restoration and/or development of peatland edge woodland on some sites, particularly to fulfil other ecological and biodiversity aims, and supported under the Forestry Grant Scheme and initiatives such as Peatland Action.

[1] http://scotland.forestry.gov.uk/supporting/strategy-policy-guidance/forestry-strategy

2.2 Peat Soil Definitions

There are slight differences in the definitions for shallow and deep peat soils used by different soil classifications, with different peat depths used as criteria to distinguish between the two, e.g. 40, 45 and 50 cm (see Kennedy (2002), FR 2010 and Appendix 1). Such small differences in peat depth are not significant for working forestry practice as peat depths are very variable in space. However, in order to be consistent in this report we refer to shallow peats with peat layer <40–50cm depth and deep peats with peat layer >40–50cm threshold. The reason for including a threshold of 40–50 cm peat depth is that various studies in the UK and wider literature have used different definitions and classifications so the depth of peat layer threshold varies. Soil C stocks are usually reported to either 15, 30, 80 or 100 cm depths. This is not coincident with the broad peat soil type classification depth criteria, but is because the stock estimate is for the total soil C to a given depth, not just that in the peat layer.

2.3 Peat Conditions in Scotland

There is currently no complete map of peatland condition. Additionally most measures of condition include vegetation and hence will assess afforested peat as degraded peat. For more details on peat condition in Scotland see Appendix 2.

2.4 Aims and Objectives

In 2010, at the request of Forestry Commission Scotland, Forest Research produced a report that examined the evidence on, and understanding of, the carbon and GHG balances of forestry on peat soils (FR 2010). The report was used to inform the development of guidance on restocking of deep peats (Forestry Commission Scotland, 2015). Since the original report, new experimental studies, soil resurveys and model application have been commissioned by Forestry Commission Scotland and Forestry Commission England and some new literature has been published. The aim of this project was to identify, evaluate and review any new relevant information and evidence since FR 2010. The project specification and detailed methodology used in this current review are reproduced in Appendix 3. The key points are summarised below, but in order to provide links to the original evidence, more extensive technical material is included in appendices as indicated.

3 New forest cover and soil carbon stock estimates on peaty soils in Scotland

- *Updated mapping of forestry (2015) on shallow peats and deep peats in Scotland has provided lower estimates of conifer areas on peat, e.g. by 22% and 17% on shallow and deep peats respectively since last forest mapping in 2001, but estimated broadleaf woodland areas have higher estimates by 32% and 4%, respectively, on shallow and deep peats.*
- *Currently, approximately 21% of the shallow peats and 17% of the deep peats areas are forested in Scotland, 620 kha and 151 kha, respectively.*
- *Data from the public Scottish forest estate shows that about 87% of conifer area planted on deep peats is on blanket bogs with half on flushed and half on unflushed bogs.*
- *Updated estimates bring total current soil C storage in forested shallow and deep peat soils in Scotland (down to 1 m depth) to 395 Mt C which is 26% higher than previously estimated and reported in FR 2010. Since FR 2010 there has been growing research focus on forest soil C stability and the factors that determine stability.*
- *Since FR 2010, a few new studies in the UK on soil C stability in mineral soils suggest that up to 70% of C in soils with high clay content and in podzolised sandy soils could be in a stable form, so*

minimal carbon loss is expected. However, there is still a lack of understanding as to what is the carbon stability and controls within shallow peaty soils.
- *The direct effects on soil C stability of afforestation, different silviculture regimes and forest management practices remain largely unknown. This is primarily due to limited understanding of the mechanisms controlling forest soil C.*

To inform estimates of the importance of peat soils in Scotland's forestry, the previous estimates of forest areas on peat in Scotland were re-assessed. In addition, new information on the various types of peat soils has been produced (for details see Appendix 4 & 5). Assessing the impact of forestry on peat soils in Scotland also requires a robust assessment of current soil C stocks and a good understanding of key factors determining the stability of soil C and the sensitivity to change. Previous soil C stock estimates were updated with the new area information and revised estimates of soil C profiles following wider sampling (see Appendices 4 & 5 for details).

4 Effects of afforestation and reforestation on peaty soil C stocks and GHG

- *Most recent evidence confirms previous findings of reductions in C stocks in the peat layer in shallow peat soils over the first 30 years of afforestation.*
- *There is some evidence that the carbon lost from peat layers through disturbance in shallow peat soils could be compensated for by the accumulation of carbon in litter and forest floor layers in the longer term under more than one rotation of forestry.*
- *Experimental results are building to provide a better estimate for the rate of C accumulation in mineral soil under peat in shallow peat soils but more studies are needed.*
- *What little new evidence there is since FR 2010 supports the suggestion that afforestation on deep peats has the largest negative impact on soil C stocks. However, there is great variability and uncertainty in such estimates.*

Since the FR 2010 review, there have been several relevant UK (n=7) and European (n=14) peer reviewed papers and some grey literature or unpublished studies (n=3). These new studies include repeat soil surveys at long term forest monitoring plots, chronosequence studies and targeted spatial soil survey resampling.

Recent resurveys of 21 peaty sites afforested during the last 40 years have shown an increase in forest floor C stock (Lilly *et al.*, 2016). Conversely, the change in soil C stock (*i.e.* that below the litter layer) suggested a loss but it was not significantly different from zero.

A large study in Kielder Forest supports these results, suggesting that afforestation of Sitka spruce over two rotations on peaty gley soils has significantly increased the C stock in the forest floor layer (Appendix 6; Vanguelova *et al.*, under review in Forestry). This is the only recent UK study which evaluates peaty soil C stock changes through all the phases from first to second rotation forestry and its findings suggest that the carbon lost through leaching, oxidation and decomposition from peat layers due to the disturbance by soil preparation for afforestation, clear-felling and reforestation could be compensated for by the C accumulation in upper organic soil horizons. It was concluded that the overall influence of conifer afforestation on carbon stocks of shallow peat soils in the Kielder Forest site is neutral over two rotations. Similar results have previously been reported for Harwood Forest, Northumberland (Zerva *et al.*, 2005). It should be noted that the sites in Harwood and Kielder Forests were subject to extensive drainage and deep ploughing when the forest was established which would have accelerated decomposition. Simola *et al.* (2012) studied C inventory changes on forestry-drained peatlands in Finland by re-sampling the peat layers in 2009 at the precise locations of prior quantitative peat mass analyses during the 1980s. Expressed on an annual basis, their results indicated an average net loss of 1.5 t C ha^{-1} y^{-1} from the soil from first rotation forest. These studies highlight the importance of long term chronosequence studies when evaluating the impacts of afforestation and restocking on soil C stocks, as short-term impacts may provide misleading conclusions.

Preliminary results at 10 afforested sites in Scotland, most of them on shallow peat soils, show that 5 out of the 10 sites have significantly increased their soil C stocks ten years after planting, while 4 have not changed and one site has significantly lost C (Vanguelova, 2015, interim FR report).

When carbon is released from the peat layer during afforestation and restocking, there is evidence to suggest that some it is moved down within the soil layers, and sequestered in the mineral soil underneath (e.g. Level II site at Llyn Brianne -first rotation sites, Wales; Swain *et al.*, 2010 - second rotation sites). This suggests that afforestation and restocking on shallow peat soils with high clay content of the underlying mineral soil may offset the loss of carbon from the peat layer. This also suggests that the currently afforestation practices such as ploughing and the widespread restocking practice of mounding which results in soil inversion (mounding) after clear-felling result in soil C translocation deeper in the soil profile and in a more stable form(see Appendix 6 for more details). More evidence of the suggested soil carbon vertical translocation and stabilisation is needed.

5 Effects of cultivation techniques on forested peat soil GHG balance

- *The type of ground preparation affects the degree of soil disturbance markedly. However, quantification of the loss of carbon from peat soils during ground preparation is rarely reported in the literature.*
- *New literature confirms that drained peatland, when compared with undrained, releases more carbon dioxide, nitrous oxide and dissolved carbon (DOC), but less methane. Whilst the soil may be a substantial source of CO_2 especially for nutrient rich peatlands, the better tree growth after drainage provides a larger C sink, which is likely to provide overall net sequestration. The long-term C balances of tree bogs is unknown.*
- *Conflicting evidence has been reported with both higher and lower amounts of DOC released downstream from drained forest peatlands in comparison to other land management activities.*
- *Additional investigations since 2010 strengthen the evidence that there is water table depth threshold for methane balance: as the water table reaches within approximately 30 cm of the surface the soil switches from a sink to a source due to methane efflux under increasingly anaerobic conditions.*

Cultivation of peaty soils will inevitably result in some increase in soil C loss, at least in the short term, although quantification is very difficult, and there is little evidence. Some quantification of the soil disturbance increase with the increased intensity of ground preparations has been published. This can be from 2% of the soil disturbed by only hand planting to almost 75% disturbance when the most intensive techniques such as trench mounding, draining and destumping are used (Appendix 6, Table A6.1). If practices changed from historical deep ploughing to shallow ploughing the soil disturbance would be reduced from 35-50% to 18-28% and to 4-12% if no ploughing were practiced. Such a wide range of soil disturbance needs to be quantified to assess likely soil carbon gains or losses.

One of the largest potential C losses from peatland soils is due to the drainage for afforestation. Forested drained peatland can vary from being a GHG source to a small sink because the C uptake by the trees and understorey vegetation can balance the soil C emissions. Limited evidence from a single German site suggests that natural bog forest is a more effective CO_2 sink in the long term than drained forest.

For disturbed peatlands that are undergoing restoration the magnitude of the methane emissions will strongly influence the net GHG balance. The meta-analysis of Worrall et al. (2011) emphasised that not all modified peatlands are C or GHG sources just as not all “pristine” peatlands are net sinks of C or GHG.

There is very limited information allowing the comparison of the effect of different ground preparation practices on the C and GHG balance. In addition, most sites will experience drainage as well, so separating particular effects is rare. Current site preparation and forest management practices should follow best practice guidelines which focus on minimum disturbance. Therefore the disturbance of the peat layer at restocking will not be as high as it was in the past. Current guidance on soil cultivation is currently undergoing a rigorous review in order to produce updated guidance minimising all impacts including carbon loss (FC soil cultivation guidance, under review).

6 Effects of forest management (thinning, clear felling, brash, stump management, Continuous Cover Forestry) on GHG emissions

- *There was no new (after FR 2010) published literature on impacts from thinning operations on soil C stocks and GHG balance. Previous findings reported no impacts of thinning, however this needs to be confirmed for local Scottish and wider UK conditions.*
- *Significant negative impacts of forest harvesting activity on shallow peat soil C stocks have been reported, but these stocks recovered during the second rotation. Whilst soil C is lost from shallow peat soil after tree planting the net carbon benefit is still high due to above ground carbon accumulation.*
- *New research (after FR 2010) suggests that whole tree harvesting (e.g. brash removal) practices may be positive for soil C stocks on shallow peat soils as there may be less subsequent decomposition of soil organic matter.*
- *New experimental evidence on stump harvesting (after FR 2010) on shallow peat soils suggests a large loss of C from the peat in comparison to much smaller losses by stump harvesting carried out on mineral soils.*

6.1 Thinning and clearfelling impacts

Afforestation and conventional forest management practices include potential risks to soil disturbance through the ground preparation, thinning processes, wood extraction and final clearfelling. When such practices are on shallow and deep peats, the likely loss of carbon is much higher (FR 2010). The latter report mentions only two studies on the impacts of thinning. There is little newer information on the impacts of thinning on peaty soils which remain poorly characterised and understood.

A few more studies have investigated the impacts of clearfell (for details see Appendix 7). Higher CO_2 emissions were mainly from decomposition of tree roots, harvesting residues and dead ground vegetation. CO_2 release from the peat itself actually decreased due to a combination of raised water table at depth and drying out at the surface from increased exposure but also likely due to reduced autotropic respiration from roots and mycorrhizae. As the water table height increased following clearfell due to reduced evapotranspiration, this resulted in a substantial increase in CH_4emissions and changes in N_2O emissions from the peat. Clearfelling was associated with an increase in particulate organic matter in stream waters but dissolved organic carbon was unchanged. Overall clearfelling reduced the C stock in the shallow peat soils. However, this loss was compensated through the second rotation forestry (for details see Appendix 7).

6.2 Whole tree harvesting (brash and stump removal)

Whole-tree harvesting (WTH, the removal of brash) as opposed to conventional harvesting (CH), where only the tree stem is removed) is practiced in upland conifer plantations as a way of maximising woody biomass yields in the UK. There remains limited data, but soil assessment at the long-term experiment in Kielder in 28-year-old second rotation Sitka spruce sites after WTH (in this study only brash removal) showed no evidence that WTH decreased soil C and N stocks compared to CH where brash had been left. On the contrary there were significantly higher concentrations and stocks in the WTH sites compared with CH. This was attributed to much higher decomposition and mineralisation rates in the CH plots than in the WTH plots.

Overall, the effect of WTH may be positive for soil C stocks on organo-mineral soils and negative on mineral soil. The overall site carbon balance (soils and trees) of WTH in the long term may not be different from CH as the slower growth of the trees is balanced by carbon preserved in the peaty soils (Appendix 7, Figure A7.1) (Vanguelova *et al*., *in preparation*). This evaluation shows the importance of assessing both below-ground and above-ground C balance in forestry to guide decisions.

If management includes stump harvesting, then the impact on soil carbon will be much higher in organic soils than mineral soils due to the high physical disturbance, and subsequent loss in CO_2efflux and in DOC leaching.

7 Modelling GHG during afforestation and restocking

- *There have been few recent reports of developments in soil C models applicable to forestry C and GHG balances.*
- *Some models focus on carbon balance and CO_2 fluxes, while only a few also consider CH_4 and N_2O fluxes; model scale is also partially driven by project aims.*
- *The longer term datasets needed for development and testing of soil GHG models are very limited, and tend to be very site specific. Available datasets needs to be more consistently reported to provide good quality data for model validation.*
- *Development of a new FR soil C dynamic model is in progress, and currently has been tested at some specific sites over annual temporal scales. It has been used for LULUCF reporting and is coupled for the first time with an aboveground model capturing the changing growth of forest and carbon inputs to soils throughout the forest rotations.*
- *Future GHG models need to take account of CH_4 and N_2O in addition to CO_2 flux and C stocks.*

7.1 Current model characteristics

Since the last report, there is little evidence of major developments of existing soil carbon models, particularly those that focus on highly organic soils. For example, the most recent model description of ECOSSE was 2011 (Smith *et al.*, 2011). A recent application of ECOSSE on six European peatland sites (Abdalla *et al.*, 2014) also identified a relationship between peatland CO_2 emissions and water table depth. They concluded that drainage will increase CO_2 emission from peatlands but the model estimates did not account for forest growth and carbon sequestration potential.

7.2 Development of a soil carbon model by Forest Research

An important dynamic is the change in carbon input to the land if it changes from pasture/arable to forestry or vice-versa. Models such as ECOSSE tend to have fixed carbon input, depending on the crop type. In establishing a forest, it will take some years before a canopy closes – this transition phase is likely to be important, particularly as it changes carbon input to both litter and soil, at a time that the underlying existing soil is responding to disturbance such as ground preparation and drainage. The Forest Research model CARBINE-SCA attempts to link forest productivity and turnover of soil and litter carbon input to a model based on a forest productivity model (CARBINE), linked to a litter and soil carbon model (based on a simplified version of ECOSSE). It should be noted that this model currently focusses on carbon and CO_2, and doesn't as yet include CH_4 and N_2O. Since this system in used in LULUCF reporting of forest land, a brief description is presented in Appendix 8.

8 Concluding remarks

- Updated areas and carbon stocks of forestry on shallow and deep peats soils for Scotland have been provided for this study.
- Important new findings have been reported on the impacts of afforestation on soil carbon stocks from repeat soil surveys and chronosequence experiments. More well-controlled chronosequence studies would improve understanding of time changes in soil and forest carbon balance through a number of forest rotations. This is important for understanding the future carbon balances of Scotland's forests.
- Further development and testing of process-based soil carbon and GHG models that also take account the methane flux are necessary.
- Further development of soil carbon process models to account for both physical and chemical stabilisation mechanisms of soil carbon translocation are also required.
- Future studies of forestry on peaty soil would benefit from measuring above-ground C components, as these drive the C input into the soil and include substantial carbon stored within the tree itself.
- Better agreement between measured soil carbon and GHG balance data would support improved model development and validation.
- In future experimental soil C stock assessments could be combined with soil GHG balance assessments to quantify the short term as well as long term changes in soil C.

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10 Appendix 1 FC Soil Classification

The Forestry Commission soil classification (Kennedy, 2002, FR 2010) (see Table 1 and Table 2) 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. 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. The 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, 2014) classification of deep peats (histosols) has a threshold>40 cm.

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	1
		Basic brown earth	1d
		Upland brown earth	1u
		Podzolic brown earth	1z
	3. Podzols	Typical podzol	3
		Hardpan podzol	3m
	4. Ironpan soils	Typical ironpan soil	4
		Podzolic ironpan soil	4z
		Intergrade ironpan soil	4b
	12. Calcareous soils	Rendzina	12a
		Calcareous brown earth	12b
		Argillic brown earth	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	6
		Podzolic peaty surface-water gley	6z
	7. Surface-water gley soils	Typical surface-water gley	7
		Podzolic surface-water gley	7z
		Brown surface-water gley	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	8a
		Juncus articulatus or *acutiflorus* bog	8b
		Juncus effusus bog	8c
		Carex bog	8d
	9. *Molinia* (or flushed blanket) bogs	*Molinia, Myrica, Salix* bog	9a
		Tussocky *Molinia* bog; *Molinia, Calluna* bog	9b
		Tussocky *Molinia, Eriophorum* bog	9c
		Non-tussocky *Molinia, Eriophorum, Trichophorum* bog	9d
		Trichophorum, Calluna, Eriophorum, Molinia bog (weakly flushed)	9e
Unflushed peatlands	10. *Sphagnum* (or flat or raised) bogs	Lowland *Sphagnum* bog	10a
		Upland *Sphagnum* bog	10b
	11. *Calluna, Eriophorum, Trichophorum* (or unflushed blanket) bogs	*Calluna* blanket bog	11a
		Calluna, Eriophorum blanket bog	11b
		Trichophorum, Calluna blanket bog	11c
		Eriophorum blanket bog	11d
	14. Eroded bogs	Shallow hagged eroded bog	14
		Deeply hagged eroded bog	14h
		Pooled eroded bog	14w

11 Appendix 2 Peat conditions in Scotland

The condition of peatland across the whole of Scotland is not well characterised. 'Condition' can refer to both vegetation and the underlying peat, though obviously there is a close link between the two. A workshop held in 2009 (Chapman *et al.* 2009) identified the need to compile a peatland vegetation map for Scotland. It was recognised that site condition monitoring was carried out by SNH (Scottish Natural Heritage) but that this was restricted to designated sites such as SACs (Special Areas of Conservation), SPAs (Special Protection Areas), Ramsar sites and SSSIs (Sites of Special Scientific Interest) and not country-wide. Also there were sites that had been characterised by SEPA (Scottish Environmental Protection Agency), FR (Forest Research) and others for research purposes but these efforts were largely uncoordinated. It was reported that some time back the NCC (Nature Conservancy Council) had produced condition maps of parts of Scotland based on satellite imagery within BGS-defined areas of peat but this data was unavailable. The JNCC (Joint Nature Conservation Committee) (2011) report "Towards an assessment of the state of UK Peatlands" states that soil condition is favourable for ca. 60% of designated sites in Scotland. However, this is to be partially expected given that they are designated. In contrast, as part of the IUCN (International Union for Conservation of Nature) Commission of Inquiry on Peatlands (Bain *et al.*, 2011), Littlewood *et al.* (2010) state that only 18% of blanket bog in the British Isles is in a natural or near-natural condition, though blanket bog in Scotland is in a better condition than that further south. In another view, Lindsay *et al.* (2014)consider that very little of the UK peat bog habitat can be regarded as 'near-pristine'.

Artz *et al.* (2013) made an attempt to produce a "probability score for peatland restoration or conservation" and mapped this at the national level. This was not peatland condition, as such, but provided insight as to how successful restoration might be. The underlying decision support tool was limited by data availability. SNH have also prepared a "Carbon and Peatland 2016 map"[2], which is a consolidated spatial dataset of 'carbon rich soil, deep peat and priority peatland habitats' in Scotland derived from existing soil and vegetation data (James Hutton Institute 1:25,000 and 1:250,000 scale soil data and Land Cover Scotland 1988). Within it they delineate Class 1 soils which are likely to be of high conservation value. This is not the same as condition but might be considered to be a first approximation. Work at the James Hutton Institute is developing spatial data peat mapping that will add to the mapped restoration probability scale (Artz *et al.*, 2013)and the estimation of peatland condition from Sentinel-2 data is being piloted (R. Artz, pers. comm.).

In all these assessments, it is inevitable that any peatland under commercial forestry would be classified as being in a 'degraded' condition since the natural vegetation would no longer exist. The exceptions would be natural bog woodland or newly created bog edge woodland. If the condition of the underlying peat under forestry is to be considered then a different set of assessment parameters would apply. Under forestry peatland condition would consider whether the peat had been previously eroded, drained, cut-over or used for agriculture, including ploughing and fertilization, and whether some understorey still remained or natural bog vegetation was evident in the rides or other open areas that might aid recolonization and restoration. The impacts of forestry ploughing and fertilization, forestry drainage, subsidence through compaction or peat oxidation or both, irreversible drying and cracking, would all be considered.

[2] http://gateway.snh.gov.uk/natural-spaces/index.jsp

12 Appendix 3 Literature Review Methodology

This project carried out an assessment through a quick scoping review using a transparent and repeatable process as detailed in the guidance at http://nora.nerc.ac.uk/512448/. The review includes all available published and grey literature since the FR 2010 report relevant to C and GHG balance impacts of afforestation and restocking in Scotland. Additional literature representing similar conditions to Scottish forestry was consulted if it was likely to address some of the key uncertainties and gaps highlighted in the FR 2010 report and particularly those related to second and subsequent rotations, and the potential role of peatland edge woodland.

Research objectives

This systematic review methodology was selected to provide an understanding of the extent of research into the GHG flux and C balances of afforested peatland ecosystems in the UK and related geographical areas, undertaken since the FR 2010 review. This review considered three main research questions:

1) Effects of afforestation on peaty soil C stocks and losses and on GHG fluxes
2) Effects of typical thinning and clear-felling practices on peaty soil C stocks and losses and on GHG fluxes
3) Effects of typical current soil preparation practices on peaty soil C stocks, losses and GHG fluxes.

For the purposes of refining the literature search, the following definitions were placed on the search components:

- *Geographical locations*: Principally Scotland but including the whole of the UK and Ireland. Studies from similar oceanic temperate areas to UK peatlands, based upon the Koeppen-Geiger climate zones, were also investigated.
- *Language restrictions*: English language literature only.
- *Date restrictions*: Literature published from 2010 to present (September 2017).
- *Population restrictions*: Peat soils and peatlands, principally blanket bog, with a surface organic layer > 40 or 50 cm. May include results from areas with blanket bog vegetation with a surface organic layer < 40 or 50 cm if relevant or depth not specified.
- *Measurement*: Carbon or GHG flux of peat soils.

12.1 Search strategy

The searches aimed to capture a comprehensive sample of the published and unpublished literature relevant to the three research questions of this review. Accordingly, a range of different sources of information were searched in order to maximise coverage. Where possible, database searches were restricted to literature published since 2010. Additionally, unpublished ('grey') literature was obtained by Forest Research and James Hutton Institute, based on expert knowledge of existing relevant research.

12.2 Databases

The following databases were searched using the terms detailed in the search terms section:

- ISI Web of Knowledge (inc. ISI Web of Science and ISI Proceedings)
- Google Scholar (Assessments were limited to the first 200 hits returned for each search)
- DART-Europe E-theses Portal
- EThOS: UK E-Theses Online Service

12.3 Search terms

Combinations of the search terms provided in Table A3.1 were applied to databases (where * denotes a wild card that may represent zero or more characters).The search results were tested for comprehensiveness through comparison with the bibliographies of several published reviews on the subject that were identified during searching. These reviews were also screened to identify potentially missed literature.

Table A3.1. Search terms used in database searches

Habitat search terms	Management/intervention	Measurement search terms
Peat*	Afforest*	Carbon*
	Forest*	GHG*
	Thinning	Greenhouse*
	Clearfell*	Green house*
	Clear*	Flux*
	Drain*	
	Fert*	
	Plough*	
	Plo*	

12.4 Inclusion/exclusion criteria

Following selection from the initial literature search, each article was required to contain certain criteria to be considered for inclusion in this review. These criteria were as follows:

- *Location:* Peatland systems and organo-mineral soils in the UK and Ireland. Studies from similar oceanic temperate areas to UK peatlands, based upon the Koeppen-Geiger climate zones, were also included.
- *Management/intervention:* Afforestation and/or forestry management practices of peat or peat-related soils. Draining, ploughing or fertilisation of peat or peat-related soils.
- *Measurement:* Quantity of Carbon or other GHG stored in, sequestered or released from peat or peat-related soils.
- *Study type:* Any primary study including measures of Carbon or GHG storage, sequestration or release from peat or peat-related soils. Studies under laboratory conditions were excluded.
- *Article release date*: Articles were excluded which were already included in the FR 2010 review, in line with the scope of this review.

The selection and filtering process is summarised in Figure A3.1 below.

12.5 Results and critical appraisal of the literature

Searches of all online databases identified 113,927 potentially relevant titles. Of these, 53 were assessed as relevant at abstract-level screening. With the addition of grey literature and research from other sources, 23 articles were considered as relevant following full text screening.

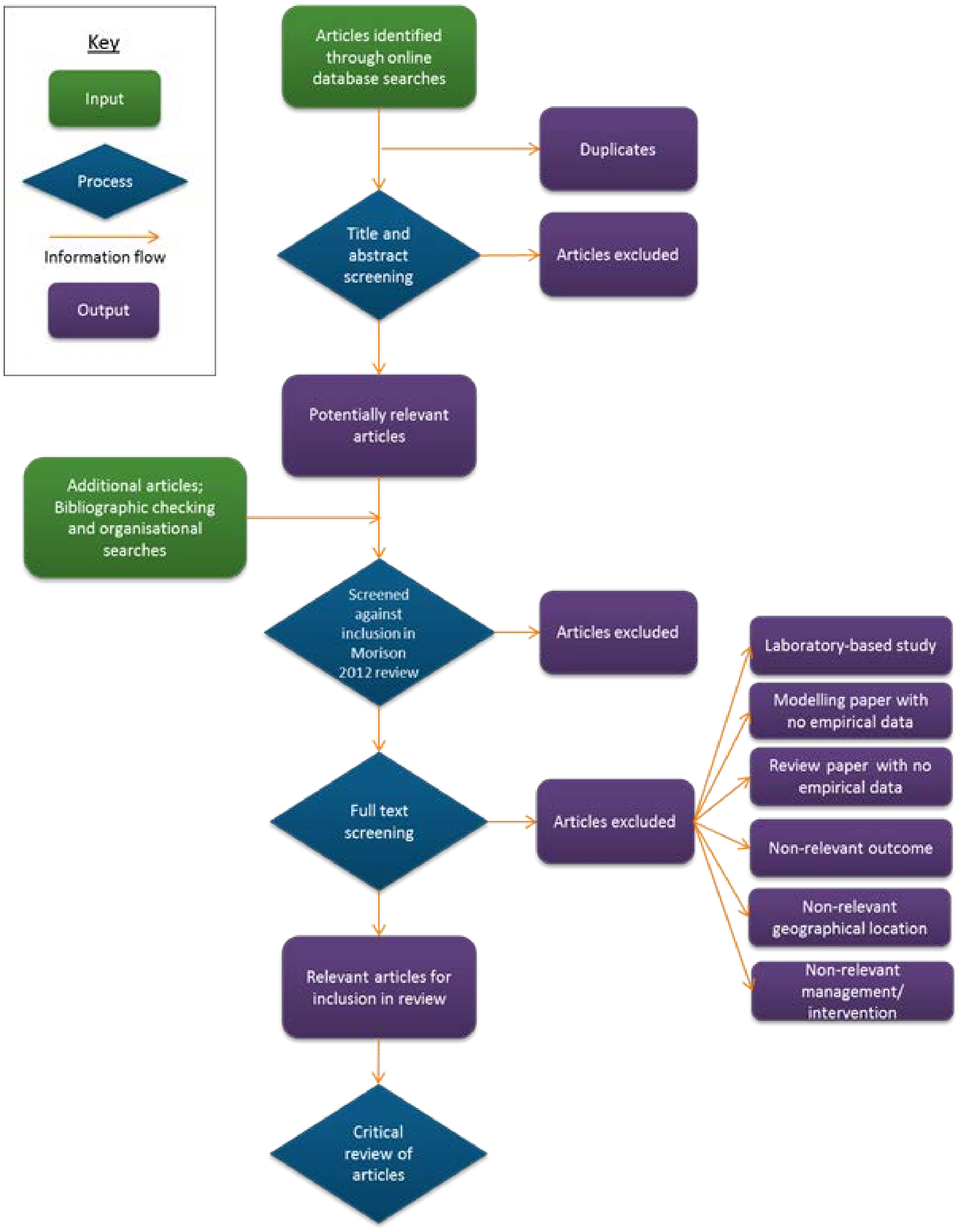

Figure A3.1 Selection and filtering scheme for the literature search.

13 Appendix 4 Forest cover and carbon stocks on peaty soils in Scotland

13.1 Forest cover on peaty soils in Scotland

The distribution of all broadleaved and conifer forests on different soil types in Scotland was updated using the most recent NFI (National Forest Inventory, 2015) woodland cover datasets, which are much more precise and capture woodlands larger than 0.5ha in comparison to the 2ha from the older National Inventory of Woodlands and Trees (NIWT) (Smith and Gilbert, 2001). The estimated total forested area on peat soils increased by 30 kha (740 to 770 kha) due in part to the increased mapping detail. Estimated conifer areas on peat have declined by 120 kha, due in part to not replanting on a proportion of deep peats coming to the end of a rotation. Broadleaf areas increased by 79 kha, particularly because of an increased area on shallow peat soils (71 kha).The two spatial datasets (NFI 2015 and James Hutton Institute 25K or 250K soil mapping) were combined and in Table 1 the areas of each woodland type on peaty soils and deep peat soils are reported. The increase in small woodland capture is evident from the data with an increase of broadleaved woodland area by 66% on peaty soils and 68% on deep peats soils, most of which will include peat edge woodland, which was not captured in the NIWT, 2001. Conifers decreased by 22% and 17% on shallow peat soils and deep peats, respectively, which is compensated for by the new broadleaf planting, mainly on the shallow peat soils (Table A4.1). The young trees category, which is about 10% of total woodland cover, includes non-high forest woodland categories (also includes coppice, low density, shrub and assumed woodland). More details of the mapping methodology, categories and terminology are listed in Appendix 5.

***Table A4.1**. Areas of each woodland canopy class on peaty and deep peat soils from NIWT 2001 and NFI 2015, and changes between the two national woodland inventories. Peaty soils include peaty gleys, peaty podzols and peaty rankers. Spatial datasets used, categories and mapping methodology is explained in Appendix 5.*

	NIWT 2001 area (km2)		NFI 2015 area (km2)			Change in area (km2)		
	Conifer	Broadleaves	Conifer	Young trees	Broadleaves	Conifer	Young trees	Broadleaves
Peaty soils	5540	369	4551	570	1075	-989	570	706
Deep peat	1452	40	1246	136	126	-206	136	86

Combining the above soil map classification with the Forestry Commission NIWT data (2001), FR 2010 calculated that coniferous forests planted on shallow and deep peats account for 48% and 13%, respectively, of the total coniferous forest area in Scotland. Broadleaved forests planted on shallow and deep peats account for only 18% and 2%, respectively, of the total broadleaved area (FR 2010). In total approximately 150 kha or 11% of Scottish forests are on deep peat soils, and 590 kha (44%) are on shallow peat soils. In our new mapping using the NFI 2015 dataset, which captured woodlands with a much smaller land area threshold (e.g. >0.5ha) compared to NIWT 2001 (>2ha), conifers planted on shallow peats and deep peats were very similar at 47 and 14% of total forested area in Scotland, respectively but broadleaf woodland area increased to 32% and 4% respectively on shallow and deep peats (Table A4.1). Currently approximately 21% of the shallow peats and 17% of deep peat areas are forested in Scotland.

13.2 Forest cover on different peat types in Scotland

The type of peatland is an important consideration in the evaluation of forestry on peat and the C stocks and GHG balance. Lowland peats differ in many respects from the blanket mires of the uplands. Whereas blanket peat bog habitat develops on areas with undulating typography and forms under conditions of high rainfall and low temperatures, lowland fens tend to form only in flat areas together with raised bogs and form under often drier and warmer conditions where natural drainage is poor. Given the differences in both the natural properties of lowland peats, and the different land uses to which they have been subjected, it is doubtful whether data obtained from studies of upland blanket bogs can be extrapolated to lowland systems and vice versa. Hence in this review, as in FR 2010 we have separated the reported findings depending on peatland types.

To investigate the forest cover on different peat types we have used the spatial data from the national forest estate in Scotland, as detailed peat type data is not available across the private forest areas. These mappings are summarised in Table A4.2 and show that about 87% of the national Scottish conifer forest planted on deep peats is on blanket bogs with half on flushed and half on unflushed bogs (for explanation of different peat types see Appendix 1). The majority of young trees, e.g. about 80%, are also on the blanket bogs, both flushed and unflushed, which are likely to consist of new planting and also the naturally occurring broadleaves and edge woodlands on deep peats, which were not captured with the previous coarser 2001 NIWT survey. The percentage of young trees out of the total national Scottish forest estate is about 10%, which is the same representation of young trees out of the total public and private forest (about 10%, see Table A4.1), so we can assume that the forestry distribution on different peat types as calculated from the public Scottish forest estate in Table A4.2 is representative for the overall forestry in Scotland.

***Table A4.2**. Area of different afforested peat types for the national forest estate according to Kennedy (2002), FC soil classification on the Scottish public forest estate using FC soil survey data (2014) (1:10K) and the NFI (2015). Equivalent data from the private sector is not available.*

Peat types (FC estate)	**Area (km^2)**		
	Conifer	*Young trees*	*Broadleaf*
Peaty gleys & peaty podzols	596	68	8
Deep peats			
Basin Bogs	25	3	1
Flushed blanket Bogs	292	31	3
Raised Bogs	57	9	1
Unflushed blanket Bog	311	21	2
Eroded Bog	10	1	0
Total deep peats	**694**	**64**	**6**

13.3 Peaty forest soils and their carbon stocks

Carbon storage in peat soils in Scotland was estimated in 2010 to be 1620 Mt C in total and 1104 Mt C in the top 1 m, with 66% of this total carbon in blanket peats, only 4% in basin peat and the rest (30%) in semi-confined peat (Chapman et al., 2010). Previous estimates of carbon stocks in forest soils (Vanguelova *et al.*, 2013) used forest cover as mapped from NIWT (2001), which can now be updated with the new area estimates from NFI.

In addition, 40 more plots have been surveyed on peaty soils (35 shallow peats, 5 on deep) since the previous forest soil survey (Vanguelova *et al.*, 2013) so updated peaty soil C stocks can also be used (Figure 1). These revised estimates as shown in Table A4.3 give a total soil carbon stock that is 26% higher than previously.

Table A4.3. Soil organic carbon stocks (in Mt C) in forested peat soils in Scotland down to 1m depth.

	conifer	*Young tree*	*broadleaf*	*total*
Deep peat	68	8	5	81
Shallow peat	231	29	54	314
total	298	36	59	395

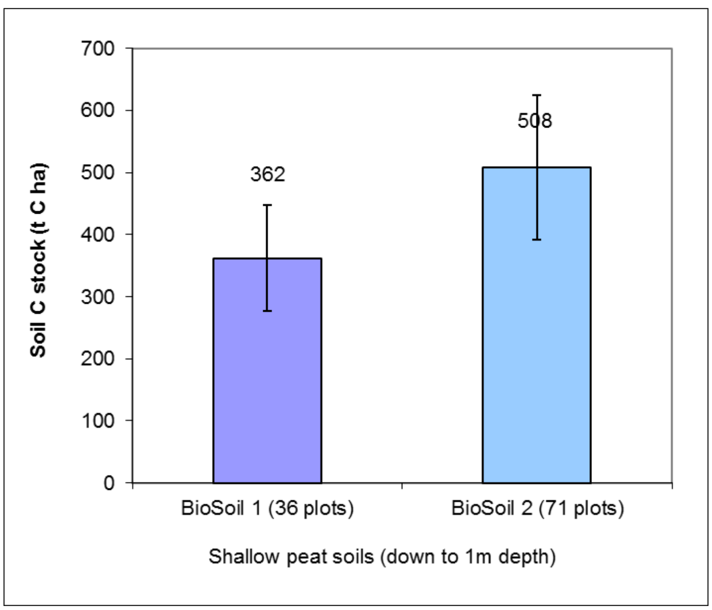

Figure A4.1. *Soil C stocks density in shallow peat soils (down to 1 m depth) assessed from Biosoil 1 from 36 plots compared to BioSoil 2 from 71 plots. Bars represent mean values and vertical bars are the standard errors of the mean.*

13.4 Carbon stability and retention in forest organic soils

Different portions of soil organic C vary in their susceptibility to microbial metabolism and thus their contribution to soil C loss. The concept of active, slow and passive C pools is widely recognized. The active C pool consists of labile material with a half-life of one or two years. The slow pool includes organic matter that is high in chemically resistant components so its life is in the order of few years or a decade. The passive pool consists of very stable materials, remaining in the soil for up to thousands of years (Brady & Weil, 2008). The proportion of different C pools in the organic matter is important for the overall risk to carbon loss or accumulation. However, if the organic matter is associated with the mineral phase of the soils, such as in shallow peat soils, then physical stabilisation processes (such as carbon binding to clay substances) could play a major role in carbon stabilization (Marschner *et al*., 2008, Schmidt *et al*., 2011). So in shallow peat soils both are important: the composition of the organic matter and the properties of the mineral soil beneath. Particularly, recent research focusses on the association between soil organic matter and the mineral phase resulting from chemical bonds which vary depending on soil mineralogical composition (Catoni *et al*., 2016).

A PhD study investigating the carbon pools in forest soils by both chemical and physical fractionation using a large subset of the BioSoil plots have found that most of the carbon (70 %) in mineral clay soils is in a very stable form (Villada *et al.*, 2013, 2016). These new findings suggest that carbon lost through leaching from peat layers due to disturbance and afforestation could be captured for long term sequestration if mineral soil under the peat layer is of heavy clay texture. In the case of peaty podzols, some of the dissolved carbon could be captured through the podzolisation process; however, it will be of more labile nature (Villada *et al.*, 2013, 2016).

In deep peat soils, the proportions of different carbon pools, and the environmental factors and management, determine the stability of the carbon. A recent study reported decreasing stable C pools and increasing rates of mineralization moving down the profile within the organic surface horizons (Lanfranchi *et al.*, submitted to Quarterly Journal of Forestry) which suggest that peat layer is more labile than litter and fermentation layers.

Afforestation as well as forest management (e.g. thinning regimes, rotation periods, brash extraction, etc.) could influence soil organic matter turnover by altering soil properties (moisture, pH, nutrient status), chemical quality of the C compounds (labile or stable) and site conditions (temperature and precipitation) thus influencing soil C sequestration (Clarke *et al.*, 2015). Advice on soil preparation which causes mixing of organic and mineral layers in shallow peat soil needs to take into consideration both the organic matter quality but also the potential for physical stabilisation of C due to mineral protection. Improved understanding of the dynamics and the stability of soil C can underpin guidance for forest practices to reduce peat C loss and aid soil C sequestration.

14 Appendix 5 Soil and woodland mapping datasets and methodology

The two spatial datasets (NFI 2015 and James Hutton Institute 25K or 250K soil mapping) were combined using the UNION tool (this creates a new layer from the two Soil maps in which the new output layer contains the combined polygons and attributes of both the original soil maps) so that both sources of information are retained in the data product. It was therefore possible to use the 25K soil data where it exists or ignore it and use the 250K soils data for the whole country. The soil and NFI 2015 data were combined using the INTERSECT tool so the soil data is clipped to the woodland area. In Appendix 4, Table 4.1 the areas of each woodland type on peaty soils and deep peat soils are reported - using soil mapping at 25K where it exists.

14.1 Forestry Commission soil survey

This dataset is the current holding of Soils data @ 31st March 2014, derived from FC soil surveys by both FC Soils Surveyors and FD staff since estimated late 1960's. Data was derived from mainly 1:10,000 scans that were georectified and digitized by Mapping staff.

The dataset was completed and unified to a recognised FC soils code with relevant conversions derived from Soil Survey reports for many maps pre-1982 (Pyatt) coding introduction. This recognised soils code matches in the table according to the attributes detailed below.

14.2 Forestry Commission NFI

The primary objective is to create a new digital map of all woodland in Great Britain using O.S. MasterMap features as boundaries where appropriate. The map shows the extent of all woodland of 0.5 ha. The NFI definition of woodland is a minimum area of 0.5 hectares under stands of trees with, or with the potential to achieve, tree crown cover of more than 20% of the ground. Areas of young trees, which have the potential to achieve a canopy cover of more than 20%, will also be interpreted as woodland and mapped. The minimum width for woodland is 20 m, although where woodlands are connected by a narrow neck of woodland less than 20 m wide, the break may be disregarded if less than 20 m in extent.

Intervening land classes are also assessed as:

Roads - all 'tarmac' roads should be excluded from the woodland area, but internal forest tracks, farmers tracks, rides etc. will be included as part of the woodland if < 20m wide.

Rivers - where the gap in woodland is 20m then rivers will be excluded from the woodland area.

Power lines etc. - where the gap in woodland is 20m then power lines will be excluded from the woodland area.

Railways - all normal gauge railways should be excluded from woodland.

Scrubby vegetation is included within this survey where low woody growth seems to dominate a likely woodland site.

The woodland boundaries have been interpreted from colour aerial orthophotographic imagery. For the base map, photographic images aimed to be no older than 3 years at the time of mapping (i.e. areas mapped in 2007 would be based on photographs that were ideally taken no earlier than 2004).Ordnance Survey MasterMap® (OSMM) features have been used as a reference for capturing the woodland boundaries. OSMM is the most up to date large-scale digital map of GB providing a seamless database for 1:1250, 1:2500 and 1:10000 survey data.

All woodland (both urban and rural, regardless of ownership) which is 0.5ha or greater in extent, with the exception of Assumed woodland or Low density areas that can be 0.1ha or greater in extend, has been mapped.

Woodland categories are defined by IFT (Interpreted Forest Type) values. Non woodland categories are defined by the IOA (Interpreted Open Area) values.

Detailed Woodland categories are:	
Broadleaved	Coppice
Conifer	Coppice with Standards
Felled	Shrub Land
Ground Prepared for New	Uncertain
Mixed –predom.	Cloud or Shadow
Mixed –predom. Conifer	Low Density
Young Trees	Assumed woodland
Failed	Windthrow/Windblow
Detailed Non woodland categories are:	
Agriculture land	River
Bare area	Road
Grass	Urban
Open water	Windfarm
Other vegetation	Quarry
Power line	

15 Appendix 6 Effects of afforestation and reforestation on peaty soil C stocks and GHG

15.1 Evidence of soil C changes

Since the FR 2010 review, there have been several relevant UK (n=7) and European (n=14) peer reviewed papers and some grey literature or unpublished studies (n=3). These new studies include repeat soil surveys at long term forest monitoring plots, chronosequence studies and targeted spatial soil survey resampling.

Recent resurveys of 21 peaty sites afforested during the last 40 years have shown an increase in forest floor carbon stock (Lilly *et al.*, 2016). The change in soil C stock (*i.e.* that below the litter layer) suggested a loss but it was not significantly different from zero. If 5 sites on deep peats were excluded from the analysis, the annual increase in overall soil profile carbon stock (including the forest floor layer) of 0.59 t C ha^{-1} y^{-1} was significant (Lilly et al., 2016).

A large study in Kielder Forest supports these results, suggesting that afforestation of Sitka spruce over two rotations on peaty gley soils has significantly increased the C stock in the forest floor layer, at an average rate of 0.73 t C ha^{-1} y^{-1} (Figure A6.1a). The overall rate of change in the peat layer C stocks was -0.35 t C ha^{-1} y^{-1} over the two forest rotations with rate of -1.8 t C ha^{-1} y^{-1} during the first half of rotation 1 changing to an increase of +0.5 to +0.8 t ha^{-1} y^{-1} during the second half of the first rotation (Vanguelova *et al.*, under review in Forestry). This is the only recent UK study which evaluates peaty soil C stock changes through all the phases from first to second rotation forestry and its findings suggest that the carbon lost through leaching, oxidation and decomposition from peat layers due to the disturbance by soil preparation for afforestation, clear-felling and reforestation could be compensated for by the C accumulation in upper organic soil horizons. The average rate of C change with afforestation over two rotations (Figure A6.1d) for the whole peaty gley profile (down to between 70 and 100 cm depth including forest floor, peat layer and mineral soil) was 0.14 t C ha^{-1} y^{-1}, varying from 0.18 to 0.54 t C ha^{-1} y^{-1}, depending on forest age and rotation. Thus, it was concluded that the overall influence of conifer afforestation on carbon stocks of shallow peat soils in the Kielder Forest site is neutral over two rotations. Simola *et al.* (2012) studied C inventory changes on forestry-drained peatlands in Finland by re-sampling the peat layers in 2009 at the precise locations of prior quantitative peat mass analyses during the 1980s. Comparison of 37 locations revealed broad variation, from slight increase to marked decrease with average losses of 74 (SE ± 25) t C ha^{-1} dry peat mass between 2009 and 1980s values. Expressed on an annual basis, their results indicated an average net loss of 1.5 t C ha^{-1} y^{-1} from the soil. The C stock change did not appear to correlate with site fertility (fertility classes according to original vegetation type), nor with post-drainage timber growth. These studies highlight the importance of long term chronosequence studies when evaluating the impacts of afforestation and restocking on soil C stocks, as short-term impacts may provide misleading conclusions.

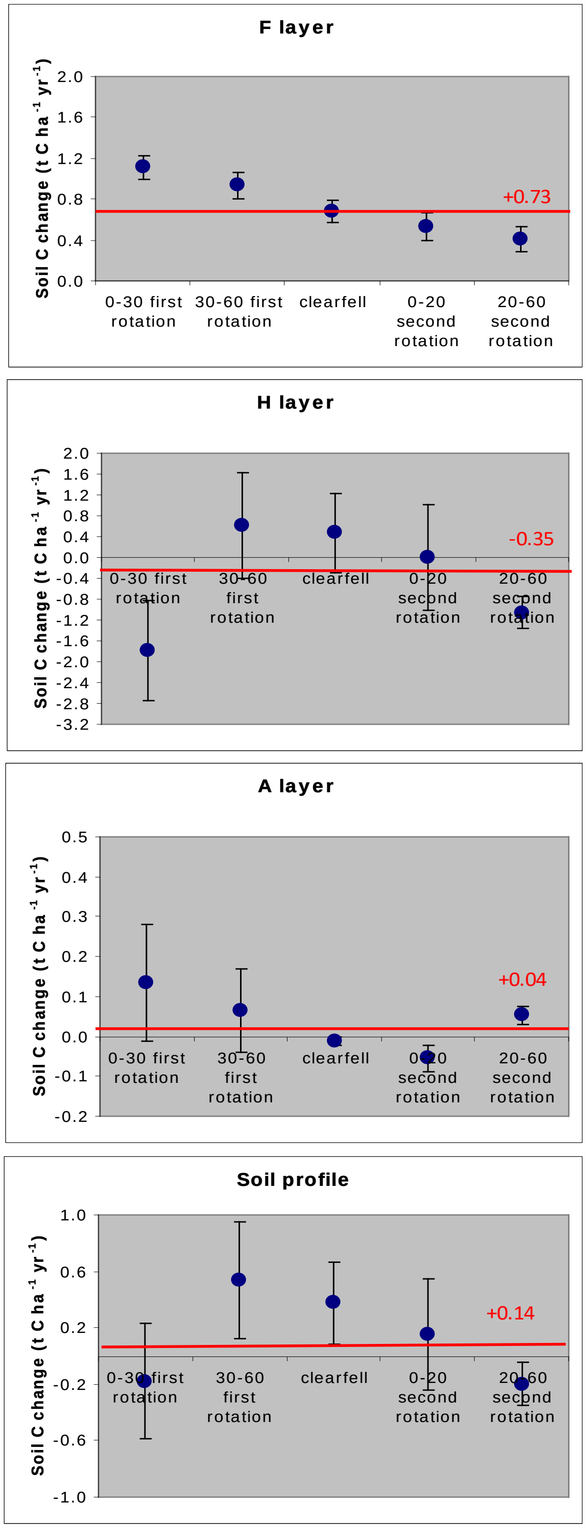

Figure A6.1. *Soil C stock change in forested plots in Kielder Forest of different ages and rotations compared to unplanted moorland, expressed as a rate over the period in the F horizon (a), peat H horizon (b), A mineral horizon (c) and total C stocks in soil profile (d). The circles represent the mean rate of C change for each age group and vertical lines are the standard errors of the mean. The horizontal red lines represent the overall average rate for each soil layer (a-c) and for the 0-1m soil profile (d).*

The Level II intensive forest monitoring plot at Coalburn in Kielder Forest, under 35 year old first rotation Sitka spruce plantation, had about 25% lower carbon stocks compared to the adjacent moorland site, e.g. C stocks down to 50 cm depth of 160 t C ha^{-1} under forest compared to 215 t C ha^{-1} for open moorland, giving a rate of loss of 2.8 t C ha^{-1} y^{-1} (Vanguelova, unpublished results). Both Harwood and Kielder forest sites in Northern England experienced heavy drainage and deep ploughing when the forest was established and these new results are similar to previous results from a chronosequence study in Harwood forest which suggested a loss of 3.3 t C ha^{-1} y^{-1} on peaty gley soil due to accelerated decomposition caused by the site preparation for drainage and planting during the afforestation process (Zerva *et al.*, 2005). A small part of this loss was compensated for by a significant increase in litter and F layer C at the rate of 0.3 t C ha^{-1} y^{-1}. Subsequently, in the second rotation there was a substantial recovery of soil C, such that including the two rotations of Sitka suggested that the C stocks had returned at the end of the second rotation to that of the original grassland.

Preliminary results are also available from a repeat soil survey from the ongoing Scottish Forest Alliance (SFA, 2003) project where afforestation was carried out over a ten-year period across 14 sites in Scotland most of them on shallow peat soils. These show that 5 out of the 10 sites analysed so far have significantly increased their C stocks in the soils, while 4 have not changed and one site has significantly lost C after tree planting (Vanguelova, 2015, interim FR report). The C stocks in the litter layer ranged from 5 - 25 t C ha^{-1} across different sites, which had developed by litter input from the growing trees and contributed to the overall site C increase at a rate of between 0.5 - 2.5 t C ha^{-1} y^{-1} for a short period of ≤10 years. The larger tree litter stocks were associated with sites which had the most significant increase in overall 0-30 cm soil C stocks. Further analysis of the baseline and complete number of SFA sites, split by soil horizon, will provide clarification of the soil C changes and likely accumulation, transport or loss of carbon.

When C is released from the peat layer through afforestation, there is evidence to suggest it is moved down within the soil layers, and sequestered in the mineral soil underneath (e.g. Swain *et al.*, 2010; Level II site at Llyn Brianne, Wales). This suggests that afforestation on shallow peat soils with high clay content of underlying mineral soil may offset the loss of carbon from the peat layer. Evidence for this vertical C translocation has been found in Llyn Brianne, an upland Sitka spruce site Level II intensive forest monitoring site on a peaty gley soil, which has lost C from the peat layer during the first rotation (at rate of 1.9 t C ha^{-1} y^{-1} over 13 years of monitoring). Significant increased DOC leaching was also detected during the last 10 years (Sawicka *et al.*, 2016), but at the same time C stocks in the clay mineral layer beneath the peat significantly increased providing a total soil profile (including litter, F layer, peat layer and mineral clay layer) C increase of about 1.5 t C ha^{-1} y^{-1} (Vanguelova, unpublished). A small increase in C stocks in the clay mineral soils under peat was detected, at the Kielder chronosequences tudy, as an effect of peat layer disturbance due to afforestation and restocking, but was not statistically significant (Figure 2b; Vanguelova *et al*., accepted in Forestry). Additional evidence for the occurrence of vertical translocation and increased stability of C with depth comes from a study investigating the disturbance of soil due to clear-felling of first rotation Sitka and planting a second rotation on peaty gley soil in Kielder forest (Swain *et al.*, 2010). This suggests that soil inversion preparation techniques (mounding) after clear-felling for the next forest rotation results in soil C translocation deeper in the soil profile and in a more stable form (see previous section on C stability).

15.2 Drainage

One of the largest potential C losses from peatland soils is due to the drainage for afforestation (Haddaway *et al.*, 2014). Forested drained peatland can vary from being a GHG source to a small sink because the C uptake by the trees and understorey vegetation can balance the soil C emissions (He *et al.*, 2016). Some key recent studies illustrated the effect of peatland drainage for forestry Pitkänen *et al.* (2013), using paired cored peat samples (0-≤ 100 cm deep) from undrained and drained sides of bog in Finland, observed marked loss of surface peat due to drainage with surface subsidence of 25-37 cm and a loss corresponding to 1.3 t C ha^{-1} y^{-1}. Yamulki *et al.* (2013) assessed the effect of tree (lodgepole pine) planting with and without intensive drainage on soil GHG fluxes after 45 years at a raised peat bog in West Flanders Moss, central Scotland. Effluxes of N_2O were low and no significant differences were observed between treatments. Annual CH_4 emissions increased with the proximity of the water table to the soil surface from 1.5 to 226.3 kg CH_4 ha^{-1} y^{-1}, respectively. For CO_2, effluxes increased by drainage from 12.3 to 16.6 t CO_2 ha^{-1} y^{-1} and dominated the total net GHG emission. Hommeltenberg *et al.*(2014) compared the CO_2 flux of a natural bog forest, and of a bog drained for forestry (Norway spruce) in southern Germany for 2 years (July 2010–June 2012). Their results indicated that the drained, forested bog was a much stronger CO_2 sink (−1.3 to −0.3 t C ha^{-1} y^{-1} in the first and second year, respectively) than the natural bog forest (−0.5 to −0.7 t C ha^{-1} y^{-1}). They explained this strong net CO_2 uptake by the high growth rates of the 44-year old spruces that exceeded the two-fold higher ecosystem respiration at the drained site. However, even though current flux measurements indicate strong CO_2 uptake of the drained spruce forest, the site was a strong net CO_2 source (13 t C ha^{-1}) over the past 44 years when the whole lifecycle since forest planting was estimated. In contrast, the natural bog-pine ecosystem has likely been a small but stable carbon sink for decades. Overall, they concluded that the natural bog forest is a more effective CO_2 sink in the long term than the drained forest in spite of a lower uptake rate during the observation period. This provides some support for the likely edge woodland positive and stable carbon balance.

Table A6.1. *Estimated disturbance of soil volume by different by intensity preparation techniques. After Bill Rayner FC soil surveyor.*

Method	Volume disturbed m3/ha	% of 0-30cm disturbed
Hand turfing, screefing	<60	2.0%
Drains at 250m/ha - 360° excavator with a draining bucket	134	4.47%
Drain mounding – 360° excavator with a drainage bucket	246	8.20%
Trench mounding + drains @250m/ha - 360° excavator	380	12.67%
Turfing – Double throw rotary mouldboard, shallow, plough	560	18.67%
Patch scarification	630	21.00%
Turfing – Double throw mouldboard, shallow, plough	710	23.67%
Disc trencher/scarifier	840	28.00%
Turfing – Double throw mouldboard, deep, plough	1,030	34.33%
Turfing – Single throw mouldboard plough	1,030	34.33%
Tine – Double throw mouldboard plough	1,430	47.67%
Tine – Single throw mouldboard plough	1,575	52.50%
Trench mounding + drains @250m/ha + de-stumping 50% area	2,232	74.40%
Agricultural ploughing	2,500	83.33%

In order to assess the GHG fluxes and the loss of carbon for nutrient rich and nutrient poor forestry-drained peatlands Ojanen *et al.* (2013)estimated the soil CO_2 balance of 68 forestry-drained (at least 20 years prior to their study) boreal peatland sites in Finland and measured fluxes of CH_4 and N_2O and the CO_2 sink of the growing tree stand. The soil was, on average, a CO_2 source of +1.9 (± 0.7) t CO_2 ha^{-1} y^{-1} at their fertile sites, but a CO_2 sink of -0.7 (± 0.3) t CO_2 ha^{-1} y^{-1} at their poor sites. The source increased at the fertile sites and the sink decreased at the poor sites with lower water table. They suggested a threshold minimum 30 cm water table depth at which the organic soil switches from a CH_4 source into a CH_4 sink and hence, this would be an optimal water table for a forestry-drained peatland, and excess drainage should be avoided. Fluxes of CH_4 and N_2O had only a minor effect, especially on the fertile sites. This is, however, true only at the site level. When considering larger areas, both fertile and poor areas co-exist. The CO_2 sink of poor sites and the CO_2 source of fertile sites will partly cancel each other out, and the relative importance of CH_4 and N_2O will become higher. The sink at the fertile sites was -6.9 ± 0.9 t CO_2eq ha^{-1} y^{-1} and at the poor sites -5.4 ± 0.7 t CO_2 eq ha^{-1} y^{-1}. The greater fertile sites' sink was due to improved tree growth, with an above ground tree stand CO_2 sink of -8.8 ± 0.6 t CO_2eq ha^{-1} y^{-1} compared to -4.9 ± 0.6 t CO_2eq ha^{-1} y^{-1} at the nutrient poor sites. They concluded that ditching-based forestry can be climatically sustainable at nutrient-poor boreal peatlands since the peat soil continues to be a CO_2 sink even after drainage. At the fertile sites, forestry will inevitably lead to loss of soil carbon in the long term, unless the tree biomass/carbon from harvest is stored as long lived products (e.g. wooden buildings, biochar in agricultural soils).

Recently, Uri *et al.,* (2017) studied the net CO_2 uptake including the soil in five downy birch (*Betula pubescens*) stands, aged between 12 and 78 years, growing on fertile well-drained histosols in eastern part of Estonia. The annual net CO_2 uptake varied with stand age accumulating 1.4–3.0 t C ha^{-1} y^{-1} in the 12–30-year-old young and middle-aged stands, zero for the 38-year-old, and was a C source of 0.95 t C ha^{-1} y^{-1} in the 78-year-old over-mature stand. Annual woody biomass increment of the stand was the main factor which determined if this forest was to act as a C accumulating system. Korkiakoski *et al.* (2017) measured exchange rates at the forest floor (tree stand were a mixture of Scots pine, Norway spruce, and downy birch) of a nutrient-rich drained peatland (fen) in Finland, which acted as a small CH_4 sink with an average of -2.19 kg CH_4 ha^{-1} y^{-1} over the 2-year measurement period. Drainage of a nutrient-poor pine bog peatland forest in southern Finland, that had been ditched about 35 years earlier, resulted in a small sink of CH_4 (-1.2 kg CH_4 ha^{-1} y^{-1}) and a small source of N_2O (1 kg N_2O ha^{-1} y^{-1}) as a result of a small (average 40 cm) drop in the water table level (Lohila *et al.*, 2011). The annual estimated net CO_2 uptake was 2.4± 0.27 t C ha^{-1} y^{-1}. This is a higher average accumulation rate than previously reported for natural northern peatlands and suggests that drainage for forestry may significantly increase the CO_2 uptake rate of nutrient-poor peatland ecosystems.

Evans *et al.* (2016) studied the composition of 'waterborne' C flux from drained organic soils and downstream oxidation of dissolved and particulate organic carbon (DOC and POC) and its potential contribution to GHG emissions in response to land-management. Overall, they estimated that waterborne C emissions may contribute about 1–4 t CO_2eq ha^{-1} y^{-1} of additional GHG emissions from drained peatlands representing around 15–50 % of total GHG emissions with the lowest emission from drained forest/semi-natural systems compared to the other land management options such as peat extraction, low or high intensively managed grassland and tropical systems. Savage (2011) reported the impacts of management practices (afforestation, drainage, grazing and burning) for upland blanket bogs in the N. Pennines on DOC release. In contrast to Evans *et al.* (2016) demonstrated that the highest DOC losses occurred from the afforested sites compared to all other management sites. However, all sites including the unmanaged site acted as DOC sources.

15.3 Preparation techniques (e.g. mounding, ploughing)

The impact on soil GHG fluxes of soil cultivation for new stand establishment on a shallow peat soil after clearcutting in central Finland was studied during a short term 22 months period by Pearson *et al.* (2012). Though annual N_2O emission levels were low (0.5–0.8 kg ha^{-1} y^{-1}), both mounding and scarifying soil preparation treatments increased the flux of N_2O from peat soil compared to the control.

When considering the fluxes of all three GHG, the cumulative impact of soil preparation (mounding or scarifying) on the global warming potential of the nutrient-poor, clearcut peatland forest was negligible compared to the control treatment. Mustamo *et al.* (2016) examined CO_2, CH_4 and N_2O emissions from a peatland complex under different land-uses in northern Finland, including grass cultivation, a Scots pine and downy birch dominated forest, a peat-extraction site and a nutrient-poor pristine mire. Soil CO_2 loss was highest in the grass site (median during growing season 30.7 t CO_2 ha^{-1} y^{-1}), and lower in the forest site and in the peat-extraction site (median for growing season 4.3 and 11.4 t CO_2 ha^{-1} y^{-1}, respectively). Both the peat-extraction site and the grass site were large sources of N_2O during the growing season (median 7.78 and 2.45 kg ha^{-1} y^{-1}, respectively) and during the winter (median 4.9 and 40.3 kg ha^{-1} y^{-1}, respectively). The pristine site was a large source of CH_4 during the growing season (median 85.8 kg ha^{-1} yr^{-1}), whereas CH_4 emissions from the drained sites during the growing season were minor. However, during winter, the peat-extraction site and the grass site emitted CH_4 (median 50.8 and 2.5 kg ha^{-1} yr^{-1}, respectively). The grass site had the highest estimated GHG emissions, due to the high CO_2 and N_2O fluxes, but the peat-extraction site also had large annual emissions, mainly as N_2O. The study suggests that raising groundwater level from 60 to 40 cm could potentially mitigate the GHG emissions from the grass site, but this would probably affect the productivity

Yamulki *et al*. (in preparation) measured soil fluxes of CO_2, CH_4 and N_2O over a 3-year period from 3 micro-topography areas (ditch, flat, mound) within two different sites in a Sitka spruce plantation on a peaty podsol at Griffin Forest in Perthshire, Scotland. The results should provide a quantitative estimate of GHG fluxes in relation to micro-topography and insight into the main soil variables affecting fluxes i.e. soil temperature, moisture and mineral N.

Current site preparation and forest management practices should follow best practice guidelines which focus on minimum disturbance. Therefore the disturbance of the peat layer at restocking will not be as high as it was in the past. Guidance on soil cultivation is currently undergoing a rigorous review in order to produce updated fit–for-purpose guidance minimising all impacts including carbon loss (FC soil cultivation guidance, under review).

15.4 Restoration effects

For disturbed peatlands methane emissions will strongly influence the net GHG balance. The meta-analysis of Worrall *et al.* (2011) reinforced that not all modified peatlands are C or GHG sources just as not all "pristine" peatlands are net sinks of C or GHG. The standard emissions factor that is used indicates a net loss for soils under forestry of 2.49 t CO_2eq ha^{-1} y^{-1}, but when the C uptake by the trees is taken into account, it is likely that the site as whole will act as a net C and GHG sink. Peatland restoration may not necessarily lead to a peatland becoming a net sink of C or GHG, because the flux of CH_4 is often a more important component of the C balance of restored peatlands. No restoration effects on soil CO_2 loss were detected in lowland peats (Haddaway *et al.*, 2014). Two studies compared drained and restored peatlands with "natural" or undrained peatlands. Both studies found negative net flux effects of -3.3+/- 7.5 (SD) t ha^{-1} y^{-1} and -8.9 +/- 9.6 (SD) t ha^{-1} y^{-1}, indicating that the drained and restored peatlands releases less CO_2 from respiration than undrained/"natural" peatlands (Soini *et al.*, 2010). Restoration resulted in an increase in CH_4 emission corresponding to 21.7 kg ha^{-1} annually and 543 kg CO_2 equivalents ha^{-1} from studies taken over 16 to 43 months (Haddaway *et al.*, 2014). No significant effects of DOC of restored versus non restored peatlands was observed in this metadata analysis.

Wilson *et al.* (2016) measured fluxes of CO_2, CH_4 and N_2O for a 5-year period at a rewetted industrial cutaway peatland in Ireland (rewetted 7 years prior to the start of the study); and compared the results with an adjacent drained area (2-year data set), and with ten long-term data sets from intact (i.e. undrained) peatlands in temperate and boreal regions. In their study N_2O emissions were not detected in either drained or rewetted sites. Rewetting reduced CO_2 emissions in un-vegetated areas by approximately 50%. When up-scaled to the ecosystem level, the emission factors (calculated as 5-year mean of annual balances) for the rewetted site were (±SD) -3.8 ± 2.9 t $CO_2$$ha^{-1}$ y^{-1} (i.e. CO_2 sink) and 120 ± 26.7kg CH_4 $ha^{-1}y^{-1}$ (i.e. CH_4 source). Nearly a decade after rewetting, the GHG balance (100-year global warming potential) had reduced noticeably (i.e. less warming) in comparison with the drained site but was still higher than comparative intact sites. Wilson *et al*. (2016) suggest that it is more likely that rewetted sites may switch from an annual CO_2 sink to a source, triggered by slightly drier conditions.

Koskinen (2017) studied the effects of restoration of forestry-drained peatlands in Finland on the nutrient and organic C exports from a catchment and to assess the differences in CH_4 emissions between undrained (pristine), drained and restored spruce swamp forests. Their results indicated potentially large effects of restoration with increased organic carbon and nutrient exports in the studied catchment and higher CH_4 emissions (192 kg CH_4 ha^{-1} y^{-1}) than in the undrained (5.5kg CH_4 ha^{-1} y^{-1}) or drained (7.7 kg CH_4 ha^{-1} y^{-1}) sites which they attributed to the restoration techniques applied.

16 Appendix 7. Effects of forest management on GHG (thinning, clear felling, brash, stump management, Continuous Cover Forestry)

16.1 Thinning and clearfelling impacts

Afforestation and conventional forest management practices include potential risks to soil disturbance through the ground preparation, thinning processes, wood extraction and final clearfelling. When such practices are on shallow and deep peats, the likely loss of carbon is much higher (FR 2010). That report mentions only two studies on the impacts of thinning. There is little further information on the impacts of thinning on peaty soils. One reason may be that the bulk of the UK forest stock is well past the time of thinning. It may also be because thinning on peat soil is not always carried out as it leads to excessive windthrow on shallow rooting forests on peat. He *et al*. (2016), modelling an afforested former agricultural fen in Sweden, found that a thinning had reduced the forest biomass by 72%. The impacts of thinning on peatland carbon and GHG exchange are poorly understood.

A few more studies have continued to look at impacts of clearfell. Mäkirantaet *al*. (2010) looked at C dynamics in the four years following clearfell in a Finnish forest. The measured CO_2 emissions were mainly from tree root decomposition, logging residue and dead ground vegetation. Respiration from the peat itself actually decreased due to a combination of raised water table at depth and drying out at the surface from increased exposure. Anderson (2010) reported on the results from two sites: where trees had been removed by whole tree harvesting the summer water table was much lower than where 'fell to recycle' had been used, with conventional harvesting being intermediate. Mäkirantaet *al*. (2012) found that the decay of logging residue following clearfell increased the carbon loss from soil organic matter; CO_2 emissions with logging residue were twice that without but were not fully accounted for by the decay of the logging residue itself. It was also observed that N_2O emissions increased with logging residues but not CH_4. Finnegan (2012) studied clearfell of coniferous forest in Ireland. She showed that the water table increased immediately following clearfell with increases in soil CO_2 loss from 11 to 19 kg C ha^{-1} d^{-1}. CH_4 emissions from peat showed a ten-fold increase but N_2O emissions decreased.

Yamulki *et al*. (in preparation) are currently measuring soil fluxes of CO_2, CH_4 and N_2O from two areas of a mature upland Sitka spruce forest on shallow peat (peaty gley) soil in Harwood Forest, Northumberland with similar previous management history. One area was harvested after one year in order to quantify the effect of clear-fell harvesting on GHG fluxes. Their preliminary results from the first 3 years showed *ca.*70% lower mean soil CO_2 efflux after felling compared to the control existing (mature) stand, probably due to reduced autotrophic respiration from roots and mycorrhizae. In contrast both CH_4 and N_2O increased by *ca.* 100 and 4 fold respectively after felling, probably due to the substantial increase in mean soil temperature and moisture and brash decomposition after felling.

A chronosequence study in Kielder forests shown that clearfelling reduced the rate of C stock change in a shallow peat soil (whole profile down to about 1m depth) from 0.54 down to 0.38 t ha^{-1} y^{-1} compared to moorland control which was contributed by reduction in soil C stocks in all soil layers, the fermentation F layer, peat layer and mineral soil layer (see Figure A6.1) (Vanguelova *et al*., accepted by Forestry). However, this loss was compensated through the second rotation forestry.

Some studies have looked at the effects of clearfell on stream waters draining peatland catchments. Rodgers *et al.* (2011) looked at stream water POC and found no effect of clearfell where correct management was implemented, involving the use of brash mats and only harvesting in dry weather. Ryder *et al.*, (2014) also looked at the impacts of clearfell on stream chemistry. They found no effect on DOC though the clearfell only accounted for <1% of the catchment area. However, they noted significant peaks in POC following their two clearfell events. O'Driscoll *et al.* (2016) reported that in-stream respiration (oxygen uptake) increased following clearfell which they ascribed to increased DOC.

16.2 Whole tree harvesting (brash and stump removal)

Whole-tree harvesting (WTH, the removal of brash) as opposed to conventional harvesting (CH), where only the tree stem is removed) is practiced often to upland conifer plantations as a way of maximising woody biomass yields in the UK. Soil assessment at the long-term experiment in Kielder in 28-year-old second rotation Sitka spruce sites after WTH (brash removal) showed no evidence that WTH decreased soil C and N stocks, but on the contrary there were significantly higher concentrations and stocks in the WTH sites compared with conventional harvesting (CH), where brash had been left. Peat gley soils contained 235 t C ha^{-1} in WTH plots compared with 160 t C ha^{-1} in the CH plots and 195 t C ha^{-1} in the fertilisation plots, all harvested in the same way after first rotation Sitka spruce (Vanguelova *et al.*, 2010). The depletion of SOC and N after CH was attributed to much higher mineralisation rates in the CH plots where brash was left on site than in the WTH plots, where significantly less soil available NO_3–N was found. These results are in accordance with extensive evidence that the retention of forest residues on site may increase the rate of mineralisation of existing soil C stocks (e.g. Mäkiranta *et al.*, 2010). At Beddgellett WTH sites, removal of logging residue did not affect the mineral soil C stocks 24 years after WTH (Walmsley and Godbold, 2010), while with WTH in Ae forest the peat layer held higher C stocks compared to conventional harvesting (Lafranchi *et al*., submitted to Quarterly Journal of Forestry). Overall, the WTH may be positive for soil carbon stocks on organo-mineral soils and negative on mineral soil. The overall site carbon balance (soils and trees) in the long term of WTH may not be changed as the slower growth of the trees is balanced by carbon preserved in the peaty soils (Figure A7.1b) (Vanguelova *et al*., *in preparation*). Although tree biomass was lowest at the WTH sites (Figure A7.1a), the total carbon balance of WTH plots of 307 t C ha^{-1} was similar to fertilisation plots with C stocks of 311 t ha^{-1} and significantly higher than conventional harvesting with 250 t C ha^{-1} (Figure A7.1b). This evaluation shows the importance of assessing both below-ground and above-ground C balance in forestry to guide decisions. However, the impacts of nutrient removal on subsequent forest growth should be accounted for to provide a complete picture of management decision impacts at the point of harvest, clearfell and restock operations.

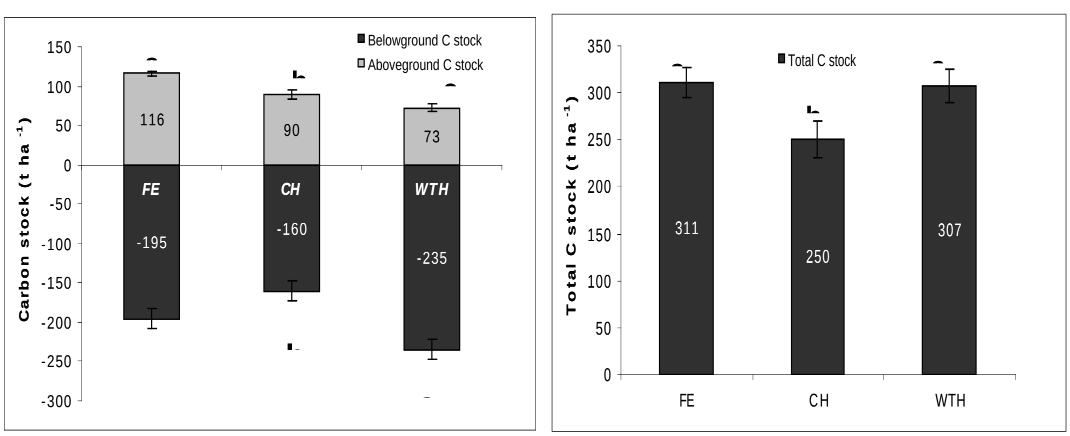

Figure A7.1. *Aboveground (trees) and belowground (soils) carbon stocks (a) and total carbon stocks (b) at the Fertilisation (FE), Conventional Harvesting (CH) and Whole Tree Harvesting (WTH) Kielder Sitka spruce plots. Solid bars are mean values and vertical bars are standard errors of the means. Different letters indicate the significant differences between carbon and nitrogen stocks between different treatments (ANOVA, p<0.05) (Vanguelova et al., in preparation).*

If management includes stump harvesting, then the impact on soil carbon will be much higher in organic soils than mineral soils due to the high physical disturbance. A study in Bala in upland Wales suggested the stump harvesting caused almost three times higher loss in soil C in peaty gley soils compared to mineral sandy brown soils (Vanguelova *et al.*, 2017). Similar results of physical soil disturbance were found in experiments in Scotland suggesting that stump harvesting followed by restock, when carried out under current guidelines, disturbed around five times the volume of soil compared to that disturbed by trench mounding (Collison *et al.*, 2015).

16.3 Less intensive forest management practices including CCF

Other less intensive management forest practices such as Continuous Cover forestry (CCF) have the potential to help adapt the forests in Scotland to some of the risks of future climate change (Stokes and Kerr, 2009). Less intensive forest management practices seek to create more diverse forests, both structurally and in terms of species composition, by avoiding clearfelling. The results from a single pilot study in Clocaenog Forest, North Wales, where about 40% of the forest is managed under CCF regime, suggest that this practice could improve forest soil quality compared to conventional forest practices (Pitman *et al.*, 2011). There was an indication that CCF could increase the stable carbon and nitrogen in the mineral soil, although differences in concentrations of C and N between the two management practices have not been observed in the organic layer (Pitman *et al.*, 2011). Establishment of research to capture the likely benefits on soil carbon and other properties by these less intensive forest management practices is vital as currently there is almost no evidence to quantify these likely benefits. CCF is widely practiced on highly fertile brown earth soils and not on peaty soils, as restructuring after canopy closure exposes a stand to increase wind risk and rooting in peaty soil is relatively poor and trees less stable. However, some CCF in Wales is also practiced on shallow peat soils.

17 Appendix 8 Modelling GHG during afforestation and restocking

17.1 Current Model Characteristics

Since the last report, there is little evidence of major developments of existing soil carbon models, particularly those that specialise or focus on highly organic soils. For example, the most recent model description of ECOSSE was 2011 (Smith *et al.*, 2011). A new version of Yasso, Yasso15 has been developed, though its description is not yet published. From the information available (e.g. Repo *et al.*, 2016) it is largely designed to operate at a large scale, relating National Forest Inventory data to climate conditions and decomposition rates. Although it can be applied at a more local scale, it doesn't take management actions such as drainage into detailed account. Increasingly there is now a dichotomy between empirical research and models which operate at different geographical and process scales. This is shown by increasing studies of short-term 'impact' events such as ground preparation (Mojeremane *et al.*, 2012) and longer term representation of land use by forestry, for example collecting chronosequence data (Vanguelova *et al.*, 2017) or re-visiting previously measured sites (Simola *et al.*, 2012). Impact studies can provide valuable insight into short term responses of soils to an activity, but fail to demonstrate the longevity of effect (for example any tapering off in emissions following drainage and ground preparation in the medium and long term). While Yasso is predominantly used at a larger geographical scale, ROMUL has recently been extended to ROMUL_HUM (Komarov *et al.*, 2017) operating at a detailed scale involving soil fauna and biota and operating at a very fine time scale. Development of models utilising such data usually lags behind data collection and validation and further improvements to models can only be carried out once sufficient data has been accumulated and processes understood. He *et al*. (2016) used the CoupModel to simulate a Norway spruce forest in southwest Sweden on fertile drained peat over 60 years, describing abiotic, biotic and GHG emissions (CO_2 and N_2O). GHG fluxes were composed of two important quantities, the spruce forest carbon (C) uptake (413 g C m^{-2} yr^{-1}) and the decomposition of peat soil (399 g C m^{-2} yr^{-1}), which nearly balanced the tree uptake. N_2O emissions contributed to the GHG emissions by up to 0.7 g N m^{-2} yr^{-1}, corresponding to 76 g C m^{-2} yr^{-1}. The 60-year old spruce forest has an accumulated biomass of 16.0 kg C m^{-2} (corresponding to 60 kg CO_2 m^{-2}). However, over this period, 26.4 kg C m^{-2} (97 kg CO_2eq m^{-2}) has been added to the atmosphere, as both CO_2 and N_2O originating from the peat soil and, indirectly, from forest thinning products. They concluded that after harvest at an age of 80 years, most of the stored biomass carbon is liable to be released, the system having captured C only temporarily and with a cost of lost peat, adding CO_2 to the atmosphere. The model results were compared with the literature in Table A8.1 below, which emphasises the few studies available, and the wide range in GHG flux values observed

***Table A8.1**. A comparison of soil peat CO_2 and N_2O emissions in the He et al. (2016) study with values published in the* literature.

Soil CO_2 flux ($g\,C\,m^{-2}\,yr^{-1}$)	Soil N_2O emissions ($g\,N\,m^{-2}\,yr^{-1}$)	Ecosystem type	Country	References
190 to 1000		Forestry-drained boreal peatland	Finland	Ojanen et al. (2013)
109 to 1200	0 to 1.9	Forest soils and other vegetated sites on deep peat	UK and other European Countries	Morison et al. (2012)
125 to 260*		Forestry-drained peatland	Finland	Minkkinen et al. (2007)
700		Grassland on agricultural fen peat	Germany	Kluge et al. (2008)
1405	1.94 (0.67)	Highly fertile drained peatland for forestry with low soil pH	Sweden	Weslien et al. (2009)
452	0.05	Afforested drained lowland raised peat bog	UK	Yamulki et al. (2013)
123 to 259*	0.02 to 0.57	Drained organic soils for deciduous and coniferous forests	Sweden	Von Arnold et al. (2005a, b)
399	0.7	Drained forested agricultural peatland agricultural peatland	Sweden	This study

* Calculated by assuming 50 % of measured soil respiration to have originated from root-based activity.

The split between highly detailed process models and less site specific but broader geographic scale is reflected in the number of parameters required to run at each scale and reflects also that improving understanding of the processes has to be balanced against national and international policy questions. Use of a highly process based model will require substantial parameterisation to get the best out of it and its extrapolation range is likely to be limited outside of the sites used for parameterisation. Conversely, a broader scale model will need to use ‘default’ parameters and fail to capture features of specific sites, but is likely to be representative of ‘average’ conditions. Clearly one of the main factors in land use change of peatlands relates to the management, and typically drainage significantly affects the aerobic/anaerobic activities within the soil, and a change in soil biota. Carlson *et al.*(2015) examine the relationship between peatland carbon loss and water table depth. This exemplifies the difficulties in modelling at a larger scale, where such dynamics will be substantially influenced by underlying strata, climate and topography. A recent application of ECOSSE on six European peatland sites (Abdalla *et al.*, 2014) also identified a relationship between peatland respiration and water table depth. They concluded that drainage will increase CO_2 emission from peatlands but did not account for forest growth and carbon sequestration potential.

17.2 Development of a soil carbon model by FR

An important dynamic is the change in carbon input to the land if it changes from pasture/arable to forestry or vice-versa. Keith *et al*. (2015) and Dondini*et al*. (2015) have reported on some effects of land use change, but on mineral soils. Models such as ECOSSE tend to have fixed carbon input, depending on the crop type. In establishing a forest, it will take some years before a canopy closes – this transition phase is likely to be important, particularly as it changes carbon input to both litter and soil, at a time that the underlying existing soil is responding to disturbance such as ground preparation and drainage. The Forest Research model CARBINE-SCA attempts to link forest productivity and turnover of soil and litter carbon input to a model based on a forest productivity model (CARBINE), and linking it to a litter and soil carbon model (based on a simplified version of ECOSSE). It should however be noted that this model currently focusses on carbon and CO_2, and doesn’t as yet include CH_4 and N_2O.Since this system in used in LULUCF reporting of forest land, a brief description is presented below and in Figure A8.6.

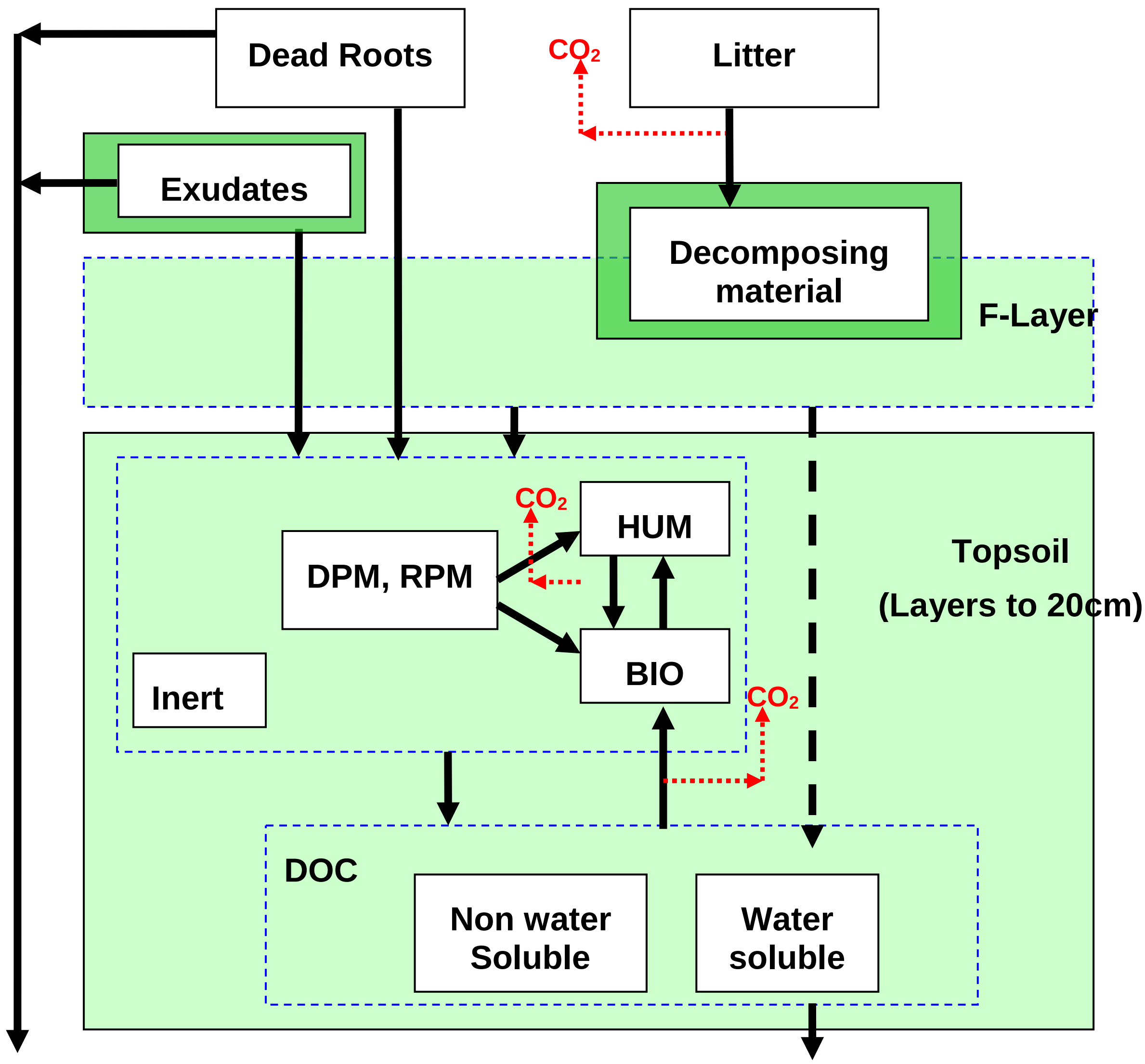

Figure A8.6. *Diagrammatic representation of the CARBINE-SCA linkages and soil pools.*

17.3 Soil carbon model challenges

Major soil carbon models such as Century, RothC, ECOSSE, Yasso, ROMUL and DNDC are built on the premise that soil organic matter can be divided into pools that have different turnover times. None of these models explicitly represents the characteristic processes of carbon transformation, such as adsorption and protection, desorption, and microbial activity. Although carbon movement between pools and their decomposition rates are modified by temperature, texture and moisture, the default turnover rates associated with individual carbon pools are justified by the combined influence of physical protection and an inferred resistance to decomposition that is dependent on substrate quality. Particularly for the ‘slow’ and ‘passive’ pools, this inherent resistance to decomposition (termed recalcitrance) has been understood to be the result of ‘humification’, with the RothC model explicitly including ‘humus’ fractions. Lack of mechanistic representation of the decomposition process produces disagreement among models and between model predictions and observational data.

For simplicity, classical biogeochemical models have generally defined soil organic matter (SOM) as a discreet number of pools differing in recalcitrance, a term mainly associated with the chemical composition of SOM, e.g. from very labile/ hydrolisable organic molecules, such as soluble, short-chained carbohydrates present in the dissolved organic fraction of SOM (DOC) to very complex, non-hydrolisable compounds such as lignin present in woody tissues of plants. Decay rates of plant litter for example, are usually inversely related to its lignin to nitrogen ratios, suggesting slow decomposition at high lignin contents (Melillo, Aber & Muratore 1982; Zhang *et al.* 2008; Prescott 2010).

As the chemical pathways in the soil are being unravelled a complex picture is slowly emerging. The classic view on humification (e.g. based on organic matter quality, chemical recalcitrant, molecular properties) remains very important under specific conditions (e.g. organic soils). There is, however, growing evidence showing that the formation of stable SOM is largely independent from its initial molecular properties (Kleber *et al.* 2011; Schmidt *et al.* 2011). In fact, it is increasingly accepted that chemical recalcitrance is only important in early stages of decomposition (von Lützow *et al.* 2006; Marschner *et al.* 2008) whereas other mechanisms, such as patterns of spatial inaccessibility against decaying soil organisms, or stabilization by interaction with mineral surfaces and metal ions (von Lützow *et al.* 2006¸ Jandl et al., 2007) might be playing a more important role in long term stabilization of SOM. Some ‘accessible’ C is not decayed for centuries while ‘recalcitrant’ pools are sometimes rapidly decomposed by specialised ectomycorrhizal fungi. Recent studies have shown that microbial products are the largest contributor to stable SOM (Cotrufo *et al.* 2013; Gleixner 2013) and this could be the case in shallow peat soils in Scotland as labile organic molecules from the above peat layer travelled through or a mixed with the mineral soil where it becomes physically stable. Some evidence from the literature stated above has already shown this may be the case in peaty gley or peaty podzol soils.

New modelling approaches need to recognize first the quality of organic compounds and its lability and accessibility to organisms and environmental factors, and second the protection of these organic compounds (e.g. in mineral complexes). The relevance of binding mechanisms of organic substances to different mineral surfaces is still uncertain (Kleber *et al.*, 2015) and the stability of minerals themselves may change as a result of exposure to organic compounds, such as those released by roots (Keiluweit *et al*., 2015). To predict the responses of soil organic carbon to climate changes or management practices, models must move beyond conceptual pools having different turnover times and instead combine soil physical principles into soil biological processes (Lehmann and Kleber, 2015). Application of this recent understanding within models poses an increasing dilemma for model developers in weighing up the detail to capture important processes, while not making relevant parameterisation impractical to apply at differing temporal and geographical scales.

18 Appendix 9 Summary of measured forest soil GHG fluxes

Table A9.1 Updated GHG fluxes reported for UK and other temperate forest soils of from standing forests on organo-mineral (OM) and deep peat sites and from clearfelled and unafforested deep peat and other vegetation sites. Negative values indicate uptake by the soil, positive values indicate emissions. Updated with additional values in red not included in Morison et al. (2012); Appendix 5, page 124. Mean values reproduced from the previous survey. Note that this table does not include C uptake by the growing trees, it is for soils only and more representing short term impacts (e.g. one rotation or less) thus the more positive emissions than negative uptake by the soil.

Activity / Site	Peat Soil type	Vegetation	N_2O (kg ha^{-1} y^{-1})	CH_4 (kg ha^{-1} y^{-1})	CO_2 (t ha^{-1} y^{-1})	References
UK forest soil						
Standing forest OM soils			1.12	9.17	20.27	
Standing forest deep peat			0.6	3.96	9.8	
East Anglia	Ditches of lowland semi-natural fen			378	11 – 14.4	Peacock et al. (2017)
Clearfelled sites OM soils			11.67	7.92	21.5	
Clearfelled sites deep peat					5.5	
Unafforested deep peatland/wetland			0.45	97.03	27.7	
Other vegetation sites OM soils			0.72	1.9	44.45	
Other vegetation sites peatland			0.19	0.775	22.7	
Europe and worldwide						
OM soils			1.89	-0.40	28.55	
Organic			4.73	113.65	16.08	
Finland	drained peatland (fen)	nutrient-rich		-1.04 to -5.05 (mean -2.19)		Korkiakoski et al. (2017)

Finland	forestry-drained peatlands	undrained (pristine) drained restored spruce swamp forests		5.5 7.8 192.7		Koskinen M. (2017)
Finland	complex peatland site under different land-use type	Forest site (soil vegintact) Forest site (soil veg removed) Pristine	0.38 -0.03 0.12	1.96 0.33 85.85	24.09 5.26 12.53	Mustamo et al. (2016)
Finland	Drained peatland dwarf-shrub pine bog forest	Nutrient poor	1	-1.2		Lohila et al. (2011)
Sweden	Drained forest agricultural peatland		11		14.64	He et al. (2016
Estonia	downy birch stands on fertile well-drained Histosols	Soil respiration heterotrophic respiration			27.1 to 32.3 17.2 to 22.7	Uri et al. (2017)