*iii* *Conversely, deforestation is the least studied of the managements that could be considered in the meta-analysis, but as for afforestation there is no complete and controlled study of peatland that is undergoing deforestation or has been deforested.*

*iv* *Given the cycle in forest plantation in the UK after the Second World War a study of both the forested state and the process of deforestation and its implications for peatland restoration should be a priority.*

*v* *Deforestation will always have to be considered in the context of the options for the use of the harvested timber, replanting or restoration to peat bog in order to give an appropriate emissions factor reflecting a life cycle analysis of the site C, even if such processes have to be accounted separately as part of inventories.*

*vi* *The research case addresses issues with regard to forested land in section 3.3.4.*

### 2.3.6 Bare and re-vegetated peatlands

In this case we consider two steady-states (bare peat soil and revegetated peat) and the transition between. It could be argued that revegetation is always a transition back to a semi-natural or undamaged state; however, given the nature of the data available it is at present worth considering as the twin of the bare peat soil

The degree of vegetation or revegetation of a peatland is the dominant control on POC flux, either through its role in limiting sediment production on intact surfaces or in reducing slope-channel linkage, i.e. breaking up the pathway eroded sediment would take before reaching a stream, in eroding but re-vegetating systems (Evans *et al* 2005). In some environments, POC removal by wind erosion is important or large peat blocks may erode downstream during stream bank collapse events but neither of these is detected by most C sampling strategies (Evans and Warburton 2001; Warburton 2003). In the UK, peat stabilisation work (e.g. using geotextiles or brash) to reduce erosion and enable revegetation usually takes place in the uplands but also the Great Fen Project and the New Forest LIFE 3 project both involve stabilisation work (www.greatfen.org.uk).

Examining the meta-analysis (Table 14) shows some unexpected results. In some cases use of lime to aid establishment of vegetation caused a decrease in soil respiration as reported by Keller *et al* (2005). In other cases we see a rise in both soil respiration and primary productivity, the rise in soil respiration is due to a rise in root respiration as the vegetation returns or due to lowering of the water table as transpiration develops. The presence of vegetation seems also to increase $CH_4$ fluxes; this could be due to increased root exudates upon the return of vegetation and/or the role of plants in transporting methane from the water table to the atmosphere. The recovery of vegetation limits soil erosion and so POC fluxes decline, but the evidence for a change in DOC is equivocal. The meta-analysis suggests that there is 70% chance of C budget improvement but only a 54% chance of GHG improvement.

**Table 14**. The summary of studies results included in the meta-analysis for revegetation of peatlands. The figures in brackets refer to the number of studies from the UK.

| | No. of studies | No. with improvement |
|---|---|---|
| NER | 8 (3) | 1 (0) |
| PP | 7 (3) | 7 (3) |
| $CH_4$ | 8 (3) | 0 (0) |
| DOC | 5 (5) | 1 (1) |
| POC | 4 (4) | 2 (2) |
| Diss. $CO_2$ | 0 (0) | 0 (0) |
| NEE | 2 (2) | 1 (1) |
| Effective sample size | C budget | 7.2 |
| | GHG | 7.7 |
| Probability of improvement | C budget | 0.77 |
| | GHG budget | 0.54 |

Rowson *et al* (in press) considered the C budget of peat covered sites on Bleaklow, Peak District, after revegetation following degradation by past wildfires and overgrazing. The study measured the C budget of eight sites: four restored-revegetated sites, two unrestored bare soil control sites, and two intact vegetated controls over two years (2006-2008). The study considered the following flux pathways: DOC; POC; dissolved $CO_2$; primary productivity; net ecosystem respiration, and $CH_4$. The study shows that bare peat sites had significant C losses as high as 522 tonnes C $km^{-2}$ $yr^{-1}$ and that revegetation resulted in an improvement in the C budgets with one revegetated site achieving a C budget after four years that was more negative than that of the vegetated control sites. The C benefit of peatland restoration in heavily degraded systems such as this may be predominantly avoided loss, i.e. the effect of revegetation is to change a large source into a small source of C or GHG rather than converting a net source into a net sink, but could be up to 833 tonnes C $km^{-2}$ $yr^{-1}$.

***Summary***

*i* *There is only one site where an emissions factor for bare soil can be derived, in which case the emissions factor would be +9.8 (±4.8)* tonnes $CO_2$eq $ha^{-1}$ $yr^{-1}$*.*

*ii* *There is only one site where an emissions factor for revegetated peatlands can be derived, in which case the emissions factor would be +5.5 (±2.4)* tonnes $CO_2$eq $ha^{-1}$ $yr^{-1}$*.*

*iii* *There is information from only one location for the derivation of these two emissions factors.*

*iv* *The meta-analysis suggests that the probability of improvement in C/ GHG flux upon revegetation is relatively low which appears to be because (a) the use of lime would appear to offset many gains of revegetation within the time frame of the available investigations, and (b) the meta-analysis technique used here may be misleading in this case, as it equally weights increases in both PP and NER that comes with re-planting yet there is no reason to believe that the increase in PP would not be considerably larger than that in NER.*

*v* *However, there is no information upon differences in restoration practise and how they may alter C or GHG budgets. It remains for future research to determine which technique would be best and to what vegetation type or cover given the need to avoid* *lime and offset any decrease in the depth of water table required to support that vegetation.*

*vi The research needs are addressed in section 3.3.5 of the research case.*

### 2.3.7 Cutover and Restored peatlands

Peat has a relatively high energy content and can therefore be used as a source of energy when combusted. Peat harvesting for fuel has been in operation throughout human history and the traditional method of hand-cutting turf and stacking blocks to dry is still in use today in some areas of Ireland and Scotland. However, the major way by which peat is commercially harvested today is milling. In this process, the surface vegetation is removed; the peat is then harrowed and dried before being sucked up by vacuum collectors. Lowland raised bog is the main source of peat for extraction (higher pH and fertility than blanket peat). Milling in the UK has been carried out to supply the horticultural market with peat for inclusion in growing media. However, in the Republic of Ireland, similarly large-scale peat extraction is used to fuel power stations. This extraction to supply the horticultural market has resulted in about 94% of lowland raised bogs in Britain being severely damaged or destroyed by peat harvesting in the 20$^{th}$ Century (Gosselink & Maltby 1990).

Peat extraction has a dramatic impact on the biodiversity of peatlands and is also a significant contributor globally to the emission of $CO_2$. Therefore, Defra is committed to preserving the stores of C in peatlands and peat soils (Defra 2007). In addition, the UK government is committed to reducing peat use under the Biodiversity Action Programme and has set targets of total market requirements for soil improvers and growing media to be supplied by non-peat materials. The target for 2005 was set at 40% and was met in that year and the very ambitious target for 2010 was set at 90%.

Restoration of cutover lowland raised bogs is occurring, such as at Thorne and Hatfield Moors, where Defra bought out the peat extraction rights in order to end commercial milling of peat at the sites in 2001. Restoration has involved re-wetting the peatland by damming the peat drains and using peat bunds to form water-retaining compartments, averaging three hectares in size, where the water level can be controlled carefully. This encourages the growth of peat forming bog species, mainly comprising *Sphagnum* mosses and *Eriophorum* sedges.

Several studies have investigated the changes in C fluxes during and after restoration of cutover raised bogs. However, as noted by Baird *et al* (2009), different components of the C balance are often reported in different papers and it can be difficult to determine whether the study site referred to in one paper is exactly the same as that reported in another. For example, Waddington and Day (2007), Waddington *et al* (2008) and Waddington *et al* (2010) report on $CH_4$ emissions, DOC fluxes and NEE, respectively, from a boreal cutover bog, Bois des Bel in Québec, Canada for one year pre-restoration (1999) and a number of years post-restoration (2000-2002). Compiling the data from each paper in Table 16 highlights that the restored site changed from a source of $CO_2$ to a small sink of $CO_2$ during the 2000 growing season and a slightly larger sink during the following year (20 tonnes C $km^{-2}$), while the cutover site remained a source of $CO_2$ during the growing season, although substantial inter-annual variability occurred. The increase in the net C sink strength at the restored site coincided with a substantial increase in vegetation cover from 22% to more than 90% in the third year following restoration. However, combining these results in the meta-analysis shows that because restoration of cutover peatlands leads to increases in $CH_4$ flux the probability of GHG benefit is restricted (Table 15).

**Table 15.** Annual fluxes (tonnes C $km^{-2}$ $yr^{-1}$) of $CO_2$, $CH_4$ and DOC from the cutover and restored areas at Bois des Bel, Quebec, Canada.

| Year | Cutover | | | Restored | | |
|---|---|---|---|---|---|---|
| | NEE | $CH_4$ | DOC | NEE | $CH_4$ | DOC |
| 1999 | 264 | -0.1 | 10.3 | 246 | 0.1 | 4.8 |
| 2000 | 137 | 0.85 | 8.5 | -10 | 0.4 | 3.4 |
| 2001 | 76 | 0.4 | 6.2 | -20 | 1.2 | 3.5 |
| 2002 | | 0.9 | | | 4.2 | |

**Table 16.** The summary of studies results included in the meta-analysis for restoration of cutover peatlands. The figures in brackets refer to the number of studies from the UK.

| | No. of studies | No. with improvement |
|---|---|---|
| NER | 0 (0) | 0 (0) |
| PP | 0 (0) | 0 (0) |
| $CH_4$ | 3 (0) | 1 (0) |
| DOC | 2 (0) | 2 (0) |
| POC | 0 (0) | 0 (0) |
| Diss. $CO_2$ | 0 (0) | 0 (0) |
| NEE | 2 (0) | 2 (0) |
| Effective sample size | C budget | 0.9 |
| | GHG | 1.7 |
| Probability of improvement | C budget | 0.42 |
| | GHG budget | 0.37 |

***Summary***

*i* *There is no reviewable data from the UK, and therefore no emissions factor can be derived.*

*ii* *Evidence from overseas suggests that $CH_4$ fluxes as a result of restoration will be problematic and may outweigh any other improvements in the GHG budget.*

*iii* *However, we would suggest that there remains a strong case for the restoration of the UK's remaining cutover peatlands to meet habitat restoration objectives and other ecosystem services as restoration of these predominantly lowland areas would otherwise be slow on a human timescale.*

*iv* *Research needs are addressed in section 3.3.6 of the research case.*

v Land uses/transitions where there is no available information on GHG and C flux

**a Cut or Mowed peatlands**

Cutting and mowing are commonly used as an alternative to burning and compared to burning cutting has the advantage of no risk of runaway wildfires or of hot burns destroying litter or soil reserves of C. The cutting or mowing of vegetation is distinct from burning in many ways:

- in that in order to save cost the cut biomass is typically left on the site, where it is likely to lead to increases in respiration;
- However,the biomass left behind can also be considered as a C input and will add to the litter layer;
- Furthermore, the presence of cut vegetation may act as mulch and keep underlying peat soils wet and help prevent surface erosion.

There is no information in Table 2 as to the spatial extent of vegetation cutting on peat soils as an intervention.

The review could not find any studies covering the effects of vegetation cutting on C or GHG budgets. As an approximation we could assume that it was similar to that of burning, i.e. vegetation removal would result in loss of biomass and cause rises in the water table but would not promote fire resistant species. Cutting is less like grazing as grazing is a slow attrition of the vegetation with vegetation still largely present whereas cutting is usually a rapid removal of vegetation by mowing or flailing to near the ground surface (as in preparation for well managed burns). It is possible therefore, we could assume that cessation of cutting would be a C benefit unless the cuttings are left and that they former a mulch and an extensive litter layer.

***Summary***

*i* *There is no information for vegetation cutting upon which to base an emissions factor or to understand the likelihood of GHG improvement.*

*ii* *If cutting becomes a serious alternative management intervention to burning then information on the impact of cutting upon the GHG and C balance will be needed.*

*iii* *The research case addresses this in section 3.3.7.*

### b Gullied and Hagged peatlands

Gullied or hagged peat represent another possible sub-class of the eroded land considered in Tables 2 through 4. Information on the spatial extent of gullied and hagged ground as opposed to bare soil has been collected as part of Natural England's survey of England's peat soils. In the absence of further information emissions factors for bare soil maybe a conservative approach; however, hagged and gullied peat maybe a more extensive type of peatland than bare soil.

***Summary***

*i* *No information exists for this land-use of for the transition from or to gullied land.*

*ii* *This issue is addressed in section 3.3.5 of the research case alongside the discussion of bare soil research.*

### c Peatlands converted to agriculture

Given the analysis in Tables 3 and 4 perhaps the most important land management that this review could not consider is that conversion to agriculture through ploughing or conversion to improved pasture. This could not be included because we found no literature on the subject. What literature was available was not for the UK and so any emissions factors would be based upon values from Bryne *et al* (2004).

***Summary***

*i It is clear that there is a complete lack of information in these settings and given the predicted emissions in Table 4 these would be a high priority for any future research programme.*

*ii The research gaps identified are addressed in section 3.3.8 of the research case.*

**d Other settings without information**

Some important peat land-use or managements have not been considered in this review. This is partly because, as above, there are no studies but also because they could be considered as being made up of elements of other land management that have been analysed in this review. For example, the impact of wind farms upon the C or GHG budgets of peat soils could not be explicitly included in this review as there are no published studies but it might be that the impact of wind farms is made of such components as drainage and the presence of bare soils.

### 2.3.8 Summary of meta-analysis

The meta-analysis is summarised for all studies in Table 17 and for UK studies only in Table 18.

**Table 17.** The summary of the meta-analysis of all studies. The figures in brackets refer to the variance in the probability estimate.

| Management | Probability of C budget improvement | Probability of GHG improvement | Effective sample (C) | Effective sample size (GHG) |
|---|---|---|---|---|
| Afforestation | 0.71 (±0.02) | 0.89 (±0.02) | 10.0 | 10.0 |
| Deforestation | 0.10 (±0.06) | 0.07 (±0.04) | 1.2 | 1.2 |
| Drainage | 0.47 (±0.01) | 0.69 (±0.02) | 9.6 | 11.0 |
| Drain-blocking | 0.57 (±0.03) | 0.33 (±0.02) | 7.2 | 8.0 |
| Grazing removal | 0.88 (±0.03) | 0.96 (±0.03) | 6.2 | 5.5 |
| Introducing Managed burning | 0.30 (±0.03) | 0.29 (±0.03) | 7.2 | 8.0 |
| Revegetation | 0.77 (±0.03) | 0.54 (±0.03) | 7.2 | 7.7 |
| Restoration of cutover peatland | 0.42 (±0.02) | 0.37 (±0.09) | 0.9 | 1.7 |

**Table 18**. The summary of the meta-analysis of UK studies. The figures in brackets refer to the variance in the probability estimate.

| Management | Probability of C budget improvement | Probability of GHG improvement | Effective sample (C) | Effective sample size (GHG) |
|---|---|---|---|---|
| Afforestation | 0.85 (±0.05) | 1.00 (±0.03) | 3.8 | 4.1 |
| Deforestation | 0.10 (±0.10) | 0.07 (±0.08) | 0.9 | 0.8 |
| Drainage | 0.58 (±0.09) | 0.66 (±0.08) | 1.7 | 2.0 |
| Drain-blocking | 0.39 (±0.06) | 0.20 (±0.07) | 1.3 | 1.6 |
| Grazing removal | 0.58 (±0.04) | 0.73 (±0.03) | 5.4 | 4.9 |
| Managed burning | 0.35 (±0.03) | 0.32 (±0.04) | 6.2 | 5.4 |
| Revegetation | 0.75 (±0.05) | 0.52 (±0.05) | 3.8 | 3.5 |
| Restoration of cutover peatland | no data | No data | 0.0 | 0.0 |

It should be noted that when no studies exist for a particular land use or management then the prior beta distribution will default to a 50% probability of improvement, i.e. it is 50:50 whether either an improvement or deterioration will be achieved; furthermore this estimate has a variance of ±50%. However, for clarity of interpretation, where there is no data that has been stated in the table (Table 18).

## 2.4 Influence of land management on C and GHG fluxes from peatlands - modelling evidence

All modelling results analysed here come from the Durham Carbon Model as described in Worrall *et al* (2009a).

### 2.4.1 'Pristine' peatlands

When considering the output of the Durham Carbon Model, the C budget of those grid squares within the modelling where there was no burning, grazing, drainage, afforestation or any other management intervention were collated. The C budget was then regressed against the altitude of the grid square; the percentage peat cover, and the percentage of bare peat soil. The resulting equation tells us about the expected C budget at any altitude.

$$C_{Total} = 0.087\,A + 1.7\,f_{peat} + 210\,f_{baresoil} - 138.5 \qquad r^2 = 96\%,\ n = 474 \qquad \text{(iii)}$$

Where: $C_{total}$ = the total C budget (tonnes C $km^{-2}$ $yr^{-1}$); A = altitude (m above sea level; asl); $f_{peat}$ = the fraction of the grid square that is peat soil; and $f_{baresoil}$ = the fraction of the grid square that is bare soil. All variables are significant at least at the 95% level and $r^2$ = 96%.

When there is 100% peat soil and no bare soil, then:

$$C_{Total} = 0.087\,A - 138.8 \qquad \text{(iv)}$$

This means that the maximum C budget that would be achieved at sea level would be -136.8 tonnes C $km^{-2}$ $yr^{-1}$ and that the average lapse rate of 8.7 tonnes C $km^{-2}$ $yr^{-1}$ 100 $m^{-1}$. The range of A for the regression is 109 to 550m asl. It should be noted that this approach predicts that there would be no *'Pristine'* peatlands that are net sources of carbon within the range of altitudes found in the UK which does not match with the observation in Table 5. However, equations (iii) - (iv) are not vegetation specific and so one of the major causes of variation in Table 5 maybe the vegetation on the site that has been taken as *'Pristine'* for that area may not be suitable for continued peat formation or expected ongoing C/GHG status, and in some cases sites in Table 5 are mature to degenerate *Calluna* which cannot be considered a peat forming species, although remains are found in peat.

### 2.4.2 Drained peatland

From the computer modelling, the presence of drainage makes a significant difference (P of no effect < 0.05) to the model outcome and decreases the C budget by -4.8 tonnes C $km^{-2}$ $yr^{-1}$ - averaged across all other conditions. When grazing is present the effect of drainage is 10.1 tonnes C $km^{-2}$ $yr^{-1}$, but when it is absent the effect is -0.6 tonnes C $km^{-2}$ $yr^{-1}$, i.e. when there is no grazing drainage could slightly improve C budgets. When burning is also present then the effect of drainage is 15.3 tonnes C $km^{-2}$ $yr^{-1}$ but when it is absent the effect of drainage is -5.8 tonnes C $km^{-2}$ $yr^{-1}$, i.e. drainage may slightly increase C budget when burning is not present. In terms of the equivalent GHG budget, drainage would be expected to result in a decrease in the budget due to the decline in $CH_4$ emissions and modelling suggests that drainage of a pristine peat would decrease equivalent $CO_2$ emissions by 19 tonnes $CO_2$ equivalent $km^{-2}$ $yr^{-1}$. It is beyond the remit of this review for detailed discussion of what the mechanisms behind these proposed emission factors might be.

Results from the Durham Carbon Model for drain-blocking will be the reverse of those for drainage at a steady state. Therefore, the model predicts that drain-blocking decreases the C budget by -4.8 tonnes C $km^{-2}$ $yr^{-1}$. When grazing is present the effect of drain-blocking is -10.1 tonnes C $km^{-2}$ $yr^{-1}$, when it is absent the effect is 0.6 tonnes C $km^{-2}$ $yr^{-1}$, i.e. when there is no grazing drain-blocking could slightly improve C budgets. When burning is present then the effect of drain blocking is -15.3 tonnes C $km^{-2}$ $yr^{-1}$ while when it is not present the effect of drain-blocking is 5.8 tonnes C $km^{-2}$ $yr^{-1}$, i.e. drainage may slightly increase C budget. In terms of the equivalent GHG budget drain-blocking would be expected to increase the budget due to the increases in $CH_4$ emissions, in this case an increase in emissions of 19 tonnes $CO_2$ equivalent $km^{-2}$ $yr^{-1}$. A recent study by Worrall *et al* (2009b) has suggested that in terms of reducing GHG emissions drain-blocking was only successful 20% of the time.

### 2.4.3 Burnt peatland

Output from the Durham Carbon Model shows that the presence of burning decreases C sequestration by an average of 83.4 tonnes C $km^{-2}$ $yr^{-1}$. Burning has a significant, but small interaction with both grazing and drainage. With respect to grazing, when there is no grazing the effect of burning increases to 89.4 tonnes C $km^{-2}$ $yr^{-1}$. When draining is present the effect of burning increases to 93 tonnes C $km^{-2}$ $yr^{-1}$. This first approach to modelling does not include any changes in the C stocks at the time of the burn. All types of fire will lead to a loss of biomass which may also be accompanied by the loss of C from the layers and the peat itself.

None of the studies above considered the production of char during the burn as a C flux and a means of additional GHG storage. Through a number of studies on burns it has been possible to estimate the extent of char production. Considering a wildfire in the Peak District in 2008, Clay and Worrall (in press) have shown that 13% of the pre-burn biomass survived the fire (the above model assumed 0% survival) and of this 13%, 4% was char. Given typical biomass for a shrub-dominated moorland these percentages would represent a 6.4 tonnes C $km^{-2}$ as char and 15.4 tonnes C $km^{-2}$ as standing dead biomass.

### 2.4.4 Grazed peatland

From the modelling approach, the presence of grazing decreases the C budget by -3.6 tonnes C $km^{-2}$ $yr^{-1}$. When there is drainage present the effect of drainage increases to 8.9

tonnes C km$^{-2}$ yr$^{-1}$ while when there is no drainage present the effect of grazing is only 1.1 tonnes C km$^{-2}$ yr$^{-1}$. When there is burning present the effect of grazing is 1.1 tonnes C km$^{-2}$ yr$^{-1}$ but when burning is not present then the effect of grazing is 6.1 tonnes C km$^{-2}$ yr$^{-1}$. With respect to GHGs it would be expected that grazing removal would decrease emissions.

### 2.4.5 Forested peatland

For the modelling approach to understanding the impact of afforestation we must consider the biomass separately. In a project based in the Galloway Forest[10], the average change of the C export from peat soil due to afforestation was 194 tonnes C km$^{-2}$ yr$^{-1}$ (-59 tonnes C km$^{-2}$ yr$^{-1}$ for pristine peat in the area compared to 134 tonnes C km$^{-2}$ yr$^{-1}$), i.e. planting of coniferous forest causes a transition from small C sink to large C source with respect to the soil. The C stored in the trees is critically dependent upon age and the growth rates of the trees which do vary. Again using data from the Galloway Forest it is possible to see that the maximum C sink will exist for this setting between 20 and 70 years (Figure 4), but after 80 years the forest biomass is mature. For the Galloway case where the forest biomass is included and given the stand age distribution of this particular forest area shows that the C budget has risen from an average of -59 tonnes C km$^{-2}$ yr$^{-1}$ to -253 tonnes C km$^{-2}$ yr$^{-1}$, but it should be reiterated that this advantage would reduce with time. These calculations do not include the possibility of use of harvested wood for fossil-fuel intensive product substitution, or woodfuel.

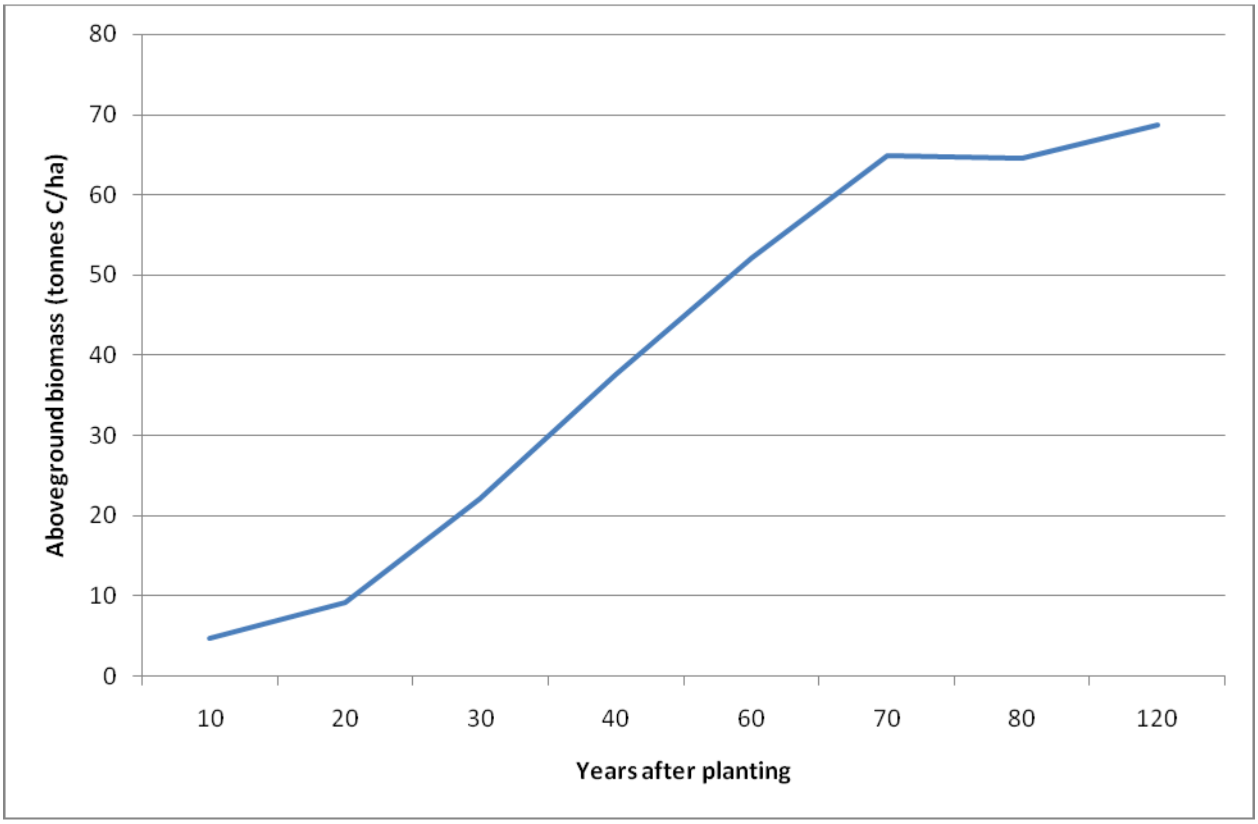


**Figure 4.** The aboveground biomass of plantation forest in Galloway over the growth of the trees.

From the modelling perspective and given the biomass curve in Figure 4 the loss of primary productivity can be predicted and there would be an optimum harvest time which in the case shown in Figure 4 would be at 80 years of age. The presence of bare soil and the role of

[10] http://homepages.see.leeds.ac.uk/~lecmsr/sustainableuplands/

revegetation are discussed later. But deforestation may form part of a product substitution programme and so if the deforestation occurred at the optimal growth stage (e.g. 70 years old - Figure 4), there was replanting and the products then used to substitute for GHG producing products then deforestation, like afforestation, could show C benefit. As an alternative, we could propose that if deforestation could occur at optimal growth stage, that the harvested wood is used for product substitution or woodfuel; that the harvested area is restored with revegetation and perhaps blocking of drainage; and that the harvested trees are replaced but planted on mineral soils then the C benefits may be maximised.

### 2.4.6 Bare and revegetated peatlands

From the modelling approach for pristine soils (Equation (iii)) the following significant relationship was found:

$$C_{Total} = 0.087A + 1.7 f_{peat} + 210 f_{baresoil} - 138.5 \qquad r^2 = 96\%,\ n = 474 \tag{v}$$

Equation (v) can be used to assess the impact of revegetation, given that equation a 1% decrease in bare soil leads to 2.1 tonnes C km$^{-2}$ yr$^{-1}$ improvement in the C budget. However, equation (iii) considers only the pristine subset within the dataset, when considering all data, i.e. including those where there was recognisable management such as burning, the equation becomes:

$$C_{Total} = 0.043A + 55.8 f_{peat} + 367.4 f_{baresoil} - 149.8 \qquad r^2 = 44\%,\ n = 4171 \tag{vi}$$

In this case the C budget lapse rate is 4.3 tonnes C km$^{-2}$ yr$^{-1}$100m$^{-1}$, and the bare soil rate has increased to 3.7 ± 0.1 tonnes C km$^{-2}$ yr$^{-1}$ % bare soil$^{-1}$. However, is there an interaction effect in which revegetation is more or less effective at greater altitude? Therefore, equation (iii) is recalculated with an interaction term $A*f_{baresoil}$ and the equation becomes:

$$C_{Total} = 0.055A + 54.8 f_{peat} + 486 f_{baresoil} - 0.27 A f_{baresoil} - 154 \qquad r^2 = 44\%,\ n = 4170 \tag{iv}$$

This would imply that the interaction although significant is slight i.e. the bare soil rate is 4.9 ± 0.4 tonnes C km$^{-2}$ yr$^{-1}$ decreases by 0.27 tonnes C km$^{-2}$ yr$^{-1}$ for every 100m decrease in altitude and so revegetation has a bigger effect at greater altitude.

### 2.4.7 Other settings

At present there are no modelling results for the restoration of cutover peat bogs or the other settings discussed above.

### 2.4.8 Summary

*i* *It is also possible to summarise the results as GHG emissions factors in contrast to those given above in Table 4 (see Table 19 & 20).*

*ii* *It should be remembered that the model used to derive this information has been developed for two sites only and has now published validation.*

*iii* *It should be a priority for research to develop, improve and validate models of the GHG balance of managed peatlands. The development of models is considered in section 5.3 of the research case.*

**Table 19.** Summary of C (not $CO_2$ eq) emission factors derived from the Durham Carbon Model.

| Management | | From modelling (tonnes C ha $yr^{-1}$) |
|---|---|---|
| Afforestation | Peat soil | +1.94[a] |
| | Above ground biomass[b] | -3.87 |
| Deforestation | | - |
| Drainage | Average | -0.05 |
| | Grazing present | +0.1 |
| | Grazing not present | -0.01 |
| | Burning present | +0.2 |
| | Burning not present | -0.06 |
| Drain-blocking | | Reverse of those above |
| Grazing removal | Average | +0.04 |
| | Drainage present | -0.09 |
| | No drainage present | -0.01 |
| | Burning present | -0.01 |
| | Burning not present | +0.06 |
| Managed burning | Average | +0.8 |
| | Grazing not present | +0.9 |
| | Drainage present | +0.9 |
| Revegetation[c] | | -3.7 |

[a] A positive number means that the carbon net sink size decrease and or the net source size increase.
[b] Assumes trees are in maximum growth phase, i.e. between 10 and 80 years old.
[c] Assumes revegetation from 100% bare soil.

**Table 20.** Summary of emission factors derived from the literature review presented above.

| Management | Emission factor (tonnes $CO_2$ eq ha $yr^{-1}$) |
|---|---|
| Pristine | +1.0 |
| Drained | +3.8 |
| Drain-blocked | +3.0 |
| Burnt | -1.4 |
| Grazed | -1.3 |
| Forested | |
| Bare | +9.8 |
| Revegetated | +5.5 |
| Cutover peatlands | - |

It should be noted that given these revised emission factors, and those given earlier (Table 3) the total GHG flux from UK peatlands would be 3.72 Mtonnes $CO_2$ eq $yr^{-1}$.

## 2.5 Influence of other factors on C and GHG fluxes from peatlands

Other factors not directly related to management can also affect C and GHG fluxes from peatlands. The most important of these are probably changes in atmospheric sulphur and nitrogen deposition and climate. This study assumes that economic changes that result in shifts in the viability of one land management over another have been considered above in the review and meta-analysis of the management impacts.

### 2.5.1 Atmospheric Deposition

Over the last century, atmospheric deposition of S over the UK increased to a peak around 1970, and then decreased dramatically; between 1986 and 2006 anthropogenic S deposition fell by 80% (RoTAP 2010). Over the same time period total N deposition has remained approximately constant, at greatly elevated levels relative to the natural baseline (Fowler *et al* 2005). The large change in S deposition, and continuing high levels of N deposition, has had several effects on the C cycle of peatlands.

Because anaerobic conditions below the peatland water table limit oxygen supply, and hence organic matter decomposition, decomposer organisms must use other electron acceptors to obtain energy from organic matter. In anaerobic ombrotrophic peatlands decomposition is usually dominated by fermenters that break down labile organic compounds to acetate, other simple organic compounds and hydrogen. Certain types of fermentation produce $CO_2$ as well as other organic carbon compounds. Depending on the availability of sulphate, these fermentation products are then used by sulphate reducers (sulphate-reducing bacteria (SRB)) or by methanogens, which transfer electrons to $SO_4^{2-}$ or $CO_2$ to produce hydrogen sulphide ($H_2S$) or methane ($CH_4$), respectively and some $CO_2$. However, in the presence of sulphate reducing bacteria the presence of sulphate will mean the suppression of methanogenesis. Thus, in peatlands where atmospheric S deposition is low, methanogenesis is the dominant anaerobic C mineralization pathway. However, S deposition on peatlands in Europe and North America increased rapidly during the 20$^{th}$ century as a result of increased emissions of sulphur dioxide ($SO_2$) from industrial combustion. This increase in $SO_4^{2-}$ deposition has the potential to enhance the importance of $SO_4^{2-}$ as an inorganic electron acceptor, and therefore divert substrate flow away from methanogenesis, thereby reducing $CH_4$ flux to the atmosphere.

A number of field and laboratory experiments have investigated the impact of $SO_4^{2-}$ deposition (1 to 15 tonnes S/km$^2$/yr, applied as either a single dose or in small, regular pulses that mimic rainfall, on $CH_4$ emissions and/or sulphate reduction rates (e.g. Dise and Verry 2001; Fowler *et al* 1995; Gauci *et al* 2002 2004a; Vile *et al* 2003; Watson and Nedwell 1998). Although the peatlands differed in their hydrological characteristics, vegetation, climate, and $CH_4$ emissions, all of these experiments showed that $SO_4^{2-}$ suppressed $CH_4$ emissions (with respect to control cores/sites) by up to *c*. 50%. Gauci *et al* (2004b) showed that a linear relationship existed between rates of atmospheric deposition up to about 3 tonnes S/km$^2$/yr and suppression of $CH_4$ emissions, but no further increase in suppression occurred with further increases in deposition. This relationship is similar to that reported by Vile *et al* (2003) between $SO_4^{2-}$ deposition and $SO_4^{2-}$ reduction rates in peatlands, which shows that sulphate-reducing bacteria are limited by something other than $SO_4^{2-}$ availability above about 2.5 tonnes S/km$^2$/yr.

Overall, the results from all the studies reviewed here strongly suggest that $SO_4^{2-}$ deposition from acid rain plays an important role in regulating $CH_4$ production in, and therefore emission from, peatlands. However, other factors have also been found to be important in controlling the extent to which $SO_4^{2-}$ inputs suppress $CH_4$ emissions. These include temperature (Gauci *et al* 2002 2004a; Nedwell and Watson 1995) position of the water table (Gauci *et al* 2002), quality of the peat organic matter, and the growth and senescence of the vascular plants. The large recorded decreases in UK sulphur deposition could result in higher rates of $CH_4$ emissions from UK peatlands. However, the response to declining inputs of atmospheric $SO_4^{2-}$ may be delayed as the sulphur retained in the peat will act as a long-term source of $SO_4^{2-}$ through cyclic reduction/oxidation due to water table fluctuations.

Flux of DOC is also affected by changes in S deposition. Over the last 20 years, concentrations of DOC have risen in freshwaters across areas of NE America and NW Europe (Skjelkvåle *et al* 2005), with particularly large and consistent increases in the UK (Evans *et al* 2005; Worrall *et al* 2004). A number of potential driving mechanisms have been proposed, many of which are linked to climate change. These include increased biological production of DOC by warming and drying (e.g. Freeman *et al* 2001a), changes in the distribution and volume of rainfall on the hydrological regime, including increasing flow volumes and changes in flow pathways (e.g. Tranvik and Jansson 2002; Erlandsson *et al* 2008; Lepistö *et al* 2008) and, increased biological activity due to elevated atmospheric $CO_2$ (Freeman *et al* 2004). While all of these factors influence DOC, Evans *et al* (2006) showed that the changes in temperature and $CO_2$ across the UK were too small to account for the observed rate of increase in DOC and that no changes in rainfall and runoff were apparent. This led Evans *et al* (2006) to suggest that the large decline in S deposition was responsible for the increase in DOC due to its impact on soil solution pH and ionic strength. The decline in S deposition has resulted in an increase in soil solution pH and decrease in ionic strength, both of which leads to an increase in the solubility of DOC. Subsequent analysis of 522 surface waters from across North America and northern Europe and found that rising trends in DOC between 1990 and 2004 can be substantially explained by changes in atmospheric deposition chemistry (Monteith *et al* 2007).

Soil acidification, due primarily to S deposition, may also suppress overall decomposition rates via the effects of increased acidity on vegetation (affecting labile substrate supply) and on decomposition processes via effects on the activity of key enzymes such as phenol oxidase (Sinsabaugh 2010). It has been suggested that this led to increased organic matter accumulation over the 'acid rain' peak of the 1960s-80s, followed by accelerated C losses as acidity has declined (Sanger *et al* 1994; Evans *et al* 2007). Recent analyses of data from the UK Countryside Survey support this hypothesis, suggesting that temporal changes in soil C are correlated with changes in soil pH (RoTAP 2010). The significance of this mechanism for peats, which are naturally acid, has yet to be determined.

Studies investigating the impact of increased N deposition (all N species) on $CH_4$ emissions have observed contrasting, and often small, impacts of N. Watson and Nedwell (1998) observed that that addition of $NO_3^-$ to peat suppressed $CH_4$ production, although this was only statistically significant at additions of 100 µM, which was much larger than $NO_3^-$ concentrations observed in the field, suggesting that $NO_3^-$ concentrations were less likely to inhibit methanogensis than $SO_4^{2-}$, and nitrate reduction would lead to $N_2O$, i.e. GHG production. Silvola *et al* (2003) investigated the effects of elevated $CO_2$ and N deposition on $CH_4$ emissions from five peatlands across Europe, including one in the UK. They observed no consistent effect of N treatment on $CH_4$ emissions across the sites, and all effects were small. In the UK site, the N-enriched and the control plots had almost the same $CH_4$ emissions. In other studies, Granberg *et al* (2001) observed that atmospheric deposition of N reduced $CH_4$ emissions from peatlands with high sedge cover but had no significant impact at sites with low sedge cover, while Aerts and de Caluwe (1999) and Nykanen *et al* (2002) both observed an increase in $CH_4$ emissions on the addition of N to nutrient-poor peatlands only.

While the direct effects of N deposition on $CH_4$ emissions may be small, the indirect effects via changes in plant community structure may be substantial (Baird *et al* 2009). Plant communities of blanket and raised bogs are adapted to nutrient-poor conditions, and the addition of increased atmospheric N deposition has the potential to result in increased growth rates, species change and loss of biodiversity. In the UK, Stevens *et al* (2004) reported a progressive loss of biodiversity along a gradient of increasing N deposition within individual habitat classes, while repeated national-scale vegetation surveys have shown a temporal shift towards more nutrient-demanding species in semi-natural upland systems (Smart *et al* 2003). Increases in N deposition have been shown to stimulate vascular plant growth at the cost of *Sphagnum*, with the magnitude of effects greatly depending on the starting conditions (Limpens *et al* 2008). Limpens *et al* (2008) suggested that the shift from a *Sphagnum*-dominated to vascular plant-dominated vegetation type would result in a general decline in C sequestration over the long-term due to increases in litter decomposability (Dorrepaal *et al* 2005) and heterotrophic respiration (Bubier *et al* 2007).

### 2.5.2 Climate change

The potential effects of climate change on C cycling in peatland systems are complex.

Climate change is being driven by increases in the concentration of atmospheric greenhouse gases, primarily $CO_2$. Increasing $CO_2$ in the atmosphere can itself lead to changes in the biogeochemistry of peat soils. Increased $CO_2$ can enhance plant growth (Freeman *et al* 2004; Silvola 1985), although some studies have observed no increase in peatland biomass growth under elevated $CO_2$ (e.g. Berendse *et al* 2001) and in others a change in plant species composition (e.g. Freeman *et al* 2004) with the abundance of vascular plants increasing relative to mosses. Freeman *et al* (2004) also observed an increase in DOC release from peat soils under elevated $CO_2$ which they attributed to elevated net primary productivity (NPP), and increased root exudation of DOC. They suggest that the labile C released by roots stimulate microbial activity, leading to enhanced degradation of soil organic matter; this process is known as the ‘priming mechanism’. Moreover, a more than additive effect was found due to the interaction between $CO_2$ and warming, leading to even greater increases in vascular plant dominance, decomposition and DOC release (Fenner *et al* 2007).

There has been much speculation on the response of peatlands to global warming, where increased temperatures are likely to lead to a decline in water tables as well as an increase in peat temperatures. Both of these are likely to lead to an increase in the decomposition of organic matter and subsequent release of $CO_2$ into the atmosphere resulting in a positive feedback to climate change. Under normal peat temperature ranges, $CO_2$ production increases by threefold for every 10°C increase, but this varies with depth and it is not clear what controls the temperature dependency of C mineralization rates (Blodau 2002). With climate change fluxes of $CH_4$ from temperate peatlands, as opposed to boreal peatlands, would be expected to decline as water tables fall and higher temperature favour $CH_4$ oxidation as much as $CH_4$ production.

Most current soil models predict that, in the longer term, rising temperatures will speed up the decomposition of organic C in soil releasing $CO_2$ into the atmosphere (Smith *et al* 2008b) in excess of any C sequestrated in the soil due to an increase in NPP (Powlson 2005), though some suggest a balance (Smith *et al* 2005 2006). However, these models are based on data derived from experiments using mineral soils, not organic soils. In addition, these models typically assume that decomposition of all soil organic matter is equally sensitive to temperature (Fang *et al* 2005). But this remains controversial (Fang *et al* 2005; Knorr *et al* 2005; Davidson & Janssens 2006). Decomposition may be slow either because the complex structures of the molecules render them resistant to decomposition or because environmental constraints restrict access of enzymes to the molecules, or a combination of the two (Davidson & Janssens 2006). The presence of high water tables in peats result in low oxygen concentrations that inhibit the activity of phenol oxidase, causing accumulation of phenolic compounds (Freeman *et al* 2001b). These phenolic compounds inhibit the activity of hydrolase enzymes responsible for decomposition, thus slowing decomposition. However, the inhibition is quickly reversed when peats become aerobic due to a drop in water table. Hence the large amount of C present in peat, which has been accumulating over centuries, is 'stable' only as long as anaerobic conditions are maintained. Once the upper layers of peat dry out they become aerobic and available for decomposition. Evidence that this process may already be occurring comes from recent repeated inventories of soils in England and Wales, which shows that peat soils lost C at a faster rate than other soil types over the last 25 years (Bellamy *et al* 2005). However, the methodology used (loss on ignition measurement of a 0-15 cm fixed depth sample) is not appropriate for measuring peat C stock change (e.g. Smith *et al* 2007) and a similar re-sampling of soil for the Countryside Survey showed no change in C content. Worrall *et al* (2007d) used the Durham Carbon Model, as described above, rather than the stocks model discussed above, in order to show that climate change over the next 30 years would mean that the C and GHG budgets of the Moor House catchment declined from net sink to net source.

Overall, most studies investigating the impact of climate change on peatland C fluxes have concentrated on gaseous exchange and the role of increased temperature and/or atmospheric $CO_2$. However, changes in rainfall and drought frequency/intensity are less discussed. Changes in water balance will lead to changes in the position of the water table. The greater the depth to the water table, the greater is the depth of oxygen ingress; this has been suggested as leading to increased flux of dissolved $CO_2$ (Jones and Mulholland 1998); demonstrated as leading to increased soil $CO_2$ respiration (Glenn *et al* 1993, Funk *et al* 1994 and Bubier *et al* 2003); and decreased $CH_4$ production and increased oxidation of the $CH_4$ being produced (Huttunen *et al* 2003).

Changes in rainfall could result in changes in runoff and river discharges which, in the UK have been tending to increase (Werrity 2002). Tranvik and Jansson (2002) have suggested that the DOC concentration increases observed by Freeman *et al* (2001a) could be hydrological changes in discharge being associated with changes in concentrations. This mechanism has been proposed for increases in DOC concentrations in lakes and stream in Sweden during the 1970s and 1980s where increases in DOC concentrations coincided with decreased temperature and increased precipitation. Similarly, a decrease in discharge as opposed to a change in the balance of hydrological pathways, can also explain increased DOC concentrations because of a decrease in dilution. But changes in precipitation can alter the balance of flowpaths in a peat-covered catchment and cause greater flow through areas rich in DOC. However, Worrall *et al* (2003) have shown that for two long-term time series, not only have there been long term increases in DOC concentration there have also been increases in DOC flux through the catchment. Alternatively, we could consider that increasing runoff is contributing to the observed increases in DOC flux. Lumsdon *et al* (2005) has proposed a model of DOC runoff concentration from organic-rich soils based upon DOC solubility, in which case increasing runoff would lead to increasing flux of DOC. Worrall and Burt (2007) have shown that the pattern of DOC flux from the UK, which is dominated by flux from peatlands, can be explained by an underlying increase in air temperature and by changes in river flow.

Rainfall is a key driver of POC flux, runoff is the primary agent of peatland erosion and increases in runoff have the potential to trigger fresh erosion through increasing the erosive force on stressed vegetation surfaces and also to exacerbate the rate of POC flux from eroding systems. The former poses much the greater risk as the shift from vegetated to eroded status entail at least an order of magnitude increase in POC flux. Changes of rate of erosion at bare sites will be of a lower order and will be significantly affected by changes in the frequency of high intensity storms which carry a large proportion of total sediment load (Evans & Warburton 2007).

There are several lines of evidence to support drought as a distinct driver of change and drought frequency is increasing in the peatlands of the UK (Worrall *et al* 2006b). Two biogeochemical mechanisms have been proposed that link drought to carbon uptake and release pathways. Freeman *et al* (2001a or b) have shown that hydrolase enzymes can be de-repressed by declines in the water table and their activity continues even after water tables rise. Alternative biogeochemical mechanism was proposed by Clark *et al* (2005). The catastrophic lowering of the water table in peat during droughts leads to the oxidation of sulphide minerals to sulphate. The increase in sulphate concentration suppresses the mobility of DOC, as the drought ends this suppression is released, as sulphate is reduced or washed out, and DOC concentrations rise. There are several lines of evidence to support these proposed mechanisms, including: observed change in the relationship between flow and DOC after a severe drought that can persist even through more minor droughts (Worrall & Burt 2004); and decoupled soil respiration and DOC production (Worrall *et al* 2005a). Alternatively, drought may alter the physical structure of peat and influence carbon release either by change of flowpath or changing water's access to parts of the peat profile, or returning to parts of the peat profile not accessible to water during the drought, including parts of the profile with increased DOC concentration. The change in the flowpaths experienced by runoff could be due to three mechanisms: the creation of new pathways; the hydrophobic nature of dried peat causing water exclusion from parts of the peat matrix; and crusting of the surface preventing infiltration. Tranvik & Jansson (2002) suggest that changes in DOC concentration in Scandinavia were due to changes in climate changing the mix of flowpaths runoff accessed. Evans *et al* (1999) have shown that runoff characteristics do change after a drought, but could only speculate on the longer term. Holden and Burt (2002) showed in laboratory studies of peat that the short term consequences of drought are to increase infiltration and flow at depth, and decrease surface runoff. However, this study showed no significant effect for drought upon mixing processes within the peat. Worrall *et al* (2006c) used a multivariate analysis of long term records of stream and soil water chemistry and showed no evidence for substantive new flowpaths being generated during the drought, though there appeared to be persistent changes in the runoff chemistry due to hydrophobic behaviour in the peat. Worrall *et al* (2007e) examined the change in runoff initiation probability over the period of the same drought and showed that runoff initiation returned to its predrought state within months of the cessation of the severe drought, i.e. flowpaths created in the drought did not persist. Therefore, studies have found only hydrophobic effects to be persistent beyond the year of the severe drought but even that is not sufficient to explain the longer term increases in DOC.

Increasing drought frequency is potentially significant for POC flux. Francis (1990) showed that desiccation during drought periods was an important process driving sediment production from bare gully walls; increases in summer drought coupled with enhanced autumn rainfalls are therefore likely to enhance POC flux. Drought conditions have also been implicated in the initiation of peat erosion in the southern Pennines by Tallis (1997 1998), Moisture stress on the surface vegetation and cracking due to desiccation have the potential to destabilise peat masses and produce a step change in POC flux from the system. A second POC related risk of drought periods relates to the risk of wildfire. Wildfires destroy vegetation creating large areas of bare peat and deep burning fires destroy roots inhibiting regeneration. During the dry summer of 1976 fire burnt over 120ha of moorland on Burbage Moor and subsequent heavy rainfall led to removal of over a metre depth of surface peat. This constitutes a catastrophic input of POC to the system and the subsequent chronic erosion of fires sites leads to long term increase in POC flux. Under conditions of increasing drought frequency the aggregation of fires scars across peatland surfaces and the associated POC losses represent a significant threat to long term peatland stability.

Future research is urgently needed to determine whether peatlands will act as a net source or sink of C in the future, particularly as over the next few decades large stock of C in organic soils could potentially be exposed to less constrained decomposition. We do not yet know the relative importance of external drivers such as climate change compared to what can be achieved by management, e.g. is that could be achieved by optimal management going to be defeated by climate change?

### 2.5.3 Summary

i *The research programme will need to reflect the influence of external (non-management) environmental influences in order to provide sufficient controls and geographic spread. Again the variation in Table 5 may in part be explained the variation in the external drivers discussed above.*

ii *The research programme could also seek to illuminate the interactions between management and the external environmental factors identified above, to help inform peatland management that is better capable of withstanding their impacts.*

iii *The need to consider the role of external drivers is considered in section 3.2.2 and section 3.2.3 of the research case.*

# 3 Conclusions

This review has highlighted that there are very few full C or GHG budget studies for peat soils in the UK, and even fewer that consider land management and land management change. Where relevant, therefore, this review has also highlighted international research findings that may provide more guidance as to potential processes and patterns of C and GHG flux under management. Many UK peatlands may be close to the tipping point between C source and C sink. Production of detailed C budgets for representative land use or management types from a wider range of peatland types would provide a clearer picture and identify systems where management intervention would have the most significant returns in terms of either reduced C loss or enhanced sequestration. This review has identified the following particular limitations:

i Measurements of some areas of peatland carbon budgets are more lacking than others: for two of the management interventions considered there is less than one equivalent complete budget, and for some management and land uses of concern we could not identify a single UK study.

ii Where emission factors could be derived they are mostly for a single site from studies that were not designed for the purpose of measuring an emissions factor.

iii If measurements of different managements and land uses are lacking then it is even more difficult to find the following:

a *Measurements relative to peatland types.* For example, although there are complete budgets of drained peat catchments there is no information relative to drainage on blanket bogs versus wet heaths or fens.

b *Measurements of managements or land-uses in different climatic settings.* As for point (a), the limited number of studies means that it is impossible to compare, for example, drain-blocking in north-east Scotland with the same intervention in the South Pennines.

c *Measurements of managements and land uses interactions.* Although this review can identify studies of individual managements or land uses, what would be the effect of multiple land uses upon one site? This review could not identify studies of whether, for example, grazing upon land subject to managed burning increases or mitigates the impact of the burning.

iv Measurements of certain components of the C or GHG budget are more lacking than others, with a particular lack of data for both $CH_4$ and $N_2O$ fluxes. The flux to the atmosphere of both these gases can be small in magnitude but they both have relatively high GWP in comparison to other gases released from peat soils.

v Most restoration transition studies have a limited time span and often lack pre-restoration monitoring of long time series (i.e. years) to generate baseline conditions prior to intervention. From what observations there are available large inter-annual variability in C, and GHG, budgets are common and thus several years of data are required to enable more informed judgements as to the C/GHG budgets under different management strategies. More focussed monitoring work is required to examine hydrological and C cycle changes following restoration using careful protocols.

vi No studies have considered budgets in a full life cycle assessment and considered issues of displacement and leakage when considering a land management change.

Such elements maybe part of other components in national GHG inventories, but will be an important part of a conservation case.

All of these limitations mean that it is not possible at this stage to greatly improve on emissions factors over those presented in this study.

While a lack of data stems partly from cost restrictions it has also been partly a function of the lack of technological capability to collect it. However, the possibility of measuring C and GHG budgets is improving rapidly with advances in technology. Automatic field logging equipment now exists, or is currently being trialled, for DOC (Grayson and Holden, in review); of dissolved $CO_2$; and for the inclusion of open-path $CH_4$ detectors on eddy flux towers. Such technology needs to be adopted in any new monitoring networks studying peatland C and GHG fluxes. The options for establishing robust, sensor-based peatland monitoring networks are considered in detail in the research case.

Despite the lack of evidence, it is possible to make the following tentative general conclusions based on the evidence compiled:

- Not all modified peatlands are C or GHG sources. Additionally, peatland restoration may not necessarily lead to a peatland becoming a C or GHG sink;

- The reason that many restoration or management interventions may not provide a benefit in terms of GHG, at least over the relatively short time which typifies most studies, is because $CH_4$ is often an important component of the C balance of restored peatlands when considered in terms of global warming potential even when, in terms of mass, $CH_4$ losses are only a few percent (3-5%) of the net exchange of $CO_2$ between the peatland and the atmosphere;

- Fluvial C fluxes also represent a significant part of the overall peatland C budget particularly in blanket bogs were large gradients can exist. In all systems the fluvial C flux will reduce the C sink associated with $CO_2$ sequestration, and in some systems they may be sufficient to change the peatland from an (apparent) C sink to a net C source. Their role in GHG terms is dependent on their ultimate fate, which remains poorly understood; and

- We can use this review to identify priorities with regard to which combinations of peatland type and or management should be regarded as priorities for further research, and this is summarised in Table 21 and 22.

The monitoring of managed peat soils is and will be expensive but is worthwhile given (i) the scale of investment in peatland restoration, conservation and management (tens to hundreds of millions of pounds each year) and ii) the importance of peatlands as a UK carbon store, a potential carbon source and a potential long-term future carbon sink. Thus a robust and comprehensive GHG and carbon monitoring programme should be properly costed, planned and implemented for the UK's peatlands. Such a programme is described in the accompanying document.

**Table 21.** Summary of review and meta-analysis of management transitions prioritised by category listing the top 5 priority land managements under each category, where 1= highest priority; 5 = lowest priority; and ? = no available information. Existing study site are listed as ✓ = possible study site exists; and× = no presently existing study site.

| **Management** | **Likely spatial extent** | **Probability of improvement** | **Effective sample size (GHG)** | **Existing study site** |
|---|---|---|---|---|
| Afforestation | 5 | 2 | | ✓ |
| Deforestation | ? | | 2 | × |
| Drainage | | 3 | 4 | ✓ |
| Drain-blocking | ? | | 3 | ✓ |
| Grazing removal | 1 | 1 | | ✓ |
| Managed burning | 4 | | | ✓ |
| Revegetation | 2 | 4 | 5 | ✓ |
| Restoration of cutover peatland | | 5 | 1 | ✓ |
| Converted for agriculture | 3 | ? | ? | ✓ |

**Table 22.** Summary of the steady-state land uses considered by the review in comparison to Table 4. The land uses are prioritised by land area and estimated emissions with the top priority under each category, where 1= highest priority for further investigation. The availability of an emissions factor for any land use is classified as ✓ = possible emissions factor could be derived from available UK literature; and ✗ = no emissions factor could be derived from available UK literature

| **Land use** | **Area (Table 2)** | **Estimated emissions (Table 4)** | **Emissions factor available** |
|---|---|---|---|
| Undamaged | 13 | | ✗ |
| Semi-natural | 1 | | ✓ |
| Cultivated | 3 | 1 | ✗ |
| Wasted | 2 | | ✗ |
| Improved grass | 6 | 2 | ✗ |
| Overgrazed | 9 | | ✗ |
| Managed burning | 4 | 3 | ✗ |
| Gripped | 5 | | ✓ |
| Eroded | 7 | | ✓ |
| Afforested | 9 | 4 | ✗ |
| Bare soil | | 5 | ✓ |
| Grazed | | | ✓ |
| Extracted | 13 | 6 | ✗ |
| Developed | 11 | | ✗ |
| Restored | 11 | 7 | ✓ |

# 4 References

*This list is only of those references used directly in the above text; studies included in the meta-analysis are listed in the appendices.*


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# Appendix 1

## Afforestation

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# Appendix 2

## Managed burning

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# Appendix 3

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# Appendix 4

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# Appendix 5

## Drain-blocking

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# Appendix 8

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