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Potential Abatement from Peatland Restoration

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Reasearch summary

Key points

- Net potential abatement benefits from peatland restoration, given our wide span of values for near-natural and damaged sites, could provide up to 9 t CO_2e ha^{-1} yr^{-1}. The figures carry large uncertainties, and in less damaged areas, much smaller emissions savings may be provided by restoration. The upper end of annual abatement range is likely to be achievable in severely damaged sites from the point that the restoration process has resulted in a near-natural state.

- Restoration of the least damaged sites may produce C savings at the lower end of this maximum figure, which may be achievable within a <10 year timeframe after restoration efforts. In addition, early intervention on such less damaged bogs also prevents further progression to a more damaged, and more highly emitting, state. Values near the upper end apply to the restoration benefits of severely damaged sites, but these will take longer to stabilize (20 to 50 years) and temporarily high methane emissions may limit early carbon savings.

- A precise figure for the area in Scotland that has the potential for some degree of change in management or active restoration is not available but is likely to be in excess of 1,000 kha of which around 350 kha may be in the 'severely damaged' category that would require active efforts to restore. These categories include previously cutover, eroded, or severely drained sites. These figures were derived from previous estimates of peatland condition in Scotland. In some cases, afforested peatlands may fall under this category, although the evidence on emissions from afforested peatlands is still equivocal.

Introduction

As will be appreciated, the provision of a figure for the potential abatement from peatland restoration is not a simple task. There are a number of factors to consider:

i. The emission factor for a particular area will depend upon the initial and restored states and this will vary depending upon the particular conditions and management regimes at each site.

ii. As many of the common land use regimes or disturbances do not manifest in a uniform fashion (e.g. grip spacings can vary), the areal extents of peatlands in different states will also vary.

iii. The timelines of abatement are also likely to vary and maximum abatement potential may not be achieved for some time following restoration.

iv. The total abatement is the integration over time of the products of emission factor and area for the various states.

v. It should be borne in mind that potential abatement may be largely made up of emission savings, i.e. a marked reduction in current carbon losses rather than a net sequestration of carbon (see Fig 7 in Bain *et al.* 2011).

vi. An additional complication is that peatlands may emit methane which has a much greater global

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warming potential in comparison to carbon dioxide. Values of CO_2e (carbon dioxide equivalents) are often calculated using a 100-year time horizon. This is a purely arbitrary period and some have argued that a 500-year time horizon is more relevant to the lifetime of a peatland and this would play down the impact of methane. Nitrous oxide emissions are generally considered to be negligible unless nitrogen fertiliser is involved or in the Central Belt where atmospheric N deposition is still a factor (Drewer *et al.,* 2010).

As requested, a detailed literature review will not be given; recent reviews have covered in detail most of the pertinent primary sources (Lindsay 2010; Worrall *et al.* 2011; Bain *et al.* 2011). What emerges is that until very recently there has been a wide range of opinions as to the efficacy of peatland restoration for significant abatement: at one end "Restoration of peatlands is a low hanging fruit, and among the most cost-effective options for mitigating climate change." (Bain *et al.* 2011), and at the other, Moran *et al.* (2011) in their computation of MACC curves concluded that "Peatland restoration may offer (a) small volume of cost-effective abatement potential but there is scientific uncertainty about the volume." and excluded it from their list of abatement measures. However, one of the authors of the latter publication has subsequently written an update in which he concludes "restoring degraded peatlands through grip blocking are comparable to some other mitigation activities currently being promoted" (Moxey 2011). Hence, a consensus appears to have been reached that peatland restoration is a highly significant carbon abatement strategy.

Additionally, it must be pointed out that current understanding is based upon very few actual studies. Only one complete budget has been published to date for a peatland in Scotland (Auchencorth Moss) and this was for a lowland raised bog in a semi-natural state (though it had been subject to some drainage in the past). Figures have been derived from partial studies at other sites in Scotland, similar studies in the rest of the UK (though again scanty) and studies in Europe and elsewhere with the caveat that the further afield from Scotland one gets the greater the differences in peatland ecology and climatic conditions and the less applicable the values obtained are likely to be.

Provisional emission factors

The available literature values have been analysed in relation to the likely condition of the peatland under study (Table 1). Data in the literature vary in terms of the completeness of the carbon budgets presented. In some cases, only carbon dioxide exchange has been studied. Net exchange of CO_2 is generally the largest component of the total C budget. Therefore, in tandem with the short timeframe available for this policy briefing, all available data on a g C m^{-2} yr^{-1} basis were converted to CO_2 equivalents using a simplistic conversion of the carbon content of carbon dioxide (i.e. multiplied by 3.664). Hence, the emission factors in Table 1 do not take into account the global warming potential of methane. However, a recent scientific discussion (February 2013) has prompted revision of this document to include a preliminary analysis for near-natural peatlands in the Northern hemisphere where the global warming potential of methane is taken into account (Table 2). The data suggest that a peatland in near-natural state is likely to function as a carbon sink, both in terms of the net ecosystem carbon balance (NECB) and also in global warming potential terms. Due to the timeframe available to produce this policy briefing, such an in-depth analysis was not feasible for the other condition classes, as this would have necessitated extensive review of the primary literature associated. In addition, although the IPCC guidelines for National Greenhouse Gas Inventories (Wetland Supplement) are currently undergoing revision, the current reporting structure under the IPCC 2006 guidelines are only based on CO_2 and N_2O emissions (IPCC, 2006).

For near-natural peatlands, the only currently available Scottish data relate to balances for Auchencorth Moss, which are between -3.7 t CO_2e ha^{-1} yr^{-1} (net uptake, Dinsmore *et al.* 2010) and +0.3 t CO_2e ha^{-1} yr^{-1} (net loss, Billett *et al.* 2004), in several climatically different years (Table 1). These figures include net exchange of greenhouse gases (CO_2 and CH_4) as well as fluvial export. Expressed in global warming terms (Table 2), this site also appears to be net cooling over all studied years.

Other data obtained from the British Isles support this Scottish figure. For blanket bogs, Koehler *et al.* (2011) studied interannual variability of greenhouse gas exchange and fluvial export in a comparable situation in an Irish blanket peat at Glencar (presumably in relatively good condition) over six consecutive years and found similar ranges of -2.4 (net uptake) to +0.3 t CO_2e ha^{-1} yr^{-1} (net loss, Table 1). In global warming terms, the site was net cooling over 4 out of the 6 studied years (Table 2). Worrall *et al.* (2009) estimated the net carbon sink at the Moorhouse NNR (a heavily impacted upland peatland in England) to vary between -0.2 to -0.71 t CO_2e ha^{-1} yr^{-1}, based on multi-annual studies of net GHG exchange and fluvial losses.

In a recent UK wide review, Billett *et al.* (2010) quote historic values of -1.3 to -7.7 t CO_2e ha^{-1} yr^{-1} based on long-term average accumulation (i.e. net increment in peat accumulation) and hence conclude that current figures are more likely in the range of -1.3 to -2.6 t CO_2e ha^{-1} yr^{-1}. The windfarm carbon calculator (Nayak *et al.* 2010) uses an example input value of -0.9 t CO_2e ha^{-1} yr^{-1} for undrained peat. Strack (2008) gives the mean long term C accumulation rate for northern bogs as -0.7 to -1.1 t CO_2e ha^{-1} yr^{-1}, again based on long term average accumulation figures, which of course integrate the net GHG and fluvial fluxes.

It is very difficult to compare some of the figures in the literature. Not all publications fully disclose how the carbon budget has been constructed, and particularly the contributions from methane emissions are often minimally discussed. In addition, very few studies take into account the uncertainty in fluvial export losses, and converting such losses to carbon dioxide equivalents is not possible as the fate of such carbon has seldom been elucidated. The only exception to this for near-natural sites are the Billett et al. 2004, Koehler et al., 2011 and Dinsmore/Drewer et al., 2010 studies, which present full carbon balances in carbon dioxide equivalents.

For damaged peatlands, the literature base on GHG emissions is particularly sparse, and, in addition, suffers from a lack of clarity on the level of degradation for each given site. Such assessments are difficult to make, because the severity of the effects on peatland functioning by drainage, afforestation, peat harvesting and other land uses are dependent on the scale of such activities and the length of time they have been in place, and the literature often does not give a complete historical account of the land management of degraded sites. Nevertheless, we have attempted to group available data on the basis of the main reported land management.

Depending on the severity of drainage activities, drained bogs may still function as net accumulators of carbon if drainage is not too severe (Table 1, full GHG equivalents for damaged sites as in Table 2 were not calculated for the purpose of this report).The only reasonably complete budget for a UK drained peatland was published by Rowson et al. (2010), where grip blocking had taken place just the year prior to measurements. Their data suggested the site to be a significant carbon source.

The effects of afforestation of peatlands are equally equivocal. Finnish research has shown that some naturally treed bogs can still be net carbon sinks, even after drainage to encourage tree growth has taken place (Lohila et al., 2011). In contrast, publications by the same group of researchers on emissions from a production forest on peatland showed that site to be net carbon emitting (Lohila et al., 2007; Table 1). The net effect appears to be dependent on the age of the tree stand, tree density, and climatic conditions. Afforestation shifts the net accumulation function from the peat soil for the most part to accumulation into tree biomass (both above-ground as timber and below-ground in root biomass). While afforested peatlands are reported to be net carbon accumulating during the early phases of tree growth, much less carbon is directed towards peat accumulation and the net benefits of carbon accumulation would be almost entirely lost from the site after tree harvest. We therefore presented the most likely range of emissions on such sites, excluding the high benefits achieved only in the early growth phase as well as the strong net emissions during the first few years following felling and restocking (e.g. Mäkiranta et al, 2010) for the most likely emission factor estimate (Table 1). The substitution effects of using timber harvested from such sites, instead of other, more carbon intensive, building materials or as fuel, may also be very substantial although it was not possible to summarise such benefits within the scope of this policy briefing. Much research is still ongoing to provide evidence of where restocking of already established forestry plantations may be viable alternative to peatland restoration; for example, in areas where peatland restoration may be extremely sensitive to current climatic conditions and where continuing tree cover would provide higher net benefits in the medium term. Minnuno et al (2010) modelled the likely CO_2 balances of peatland afforestation and concluded that 4 rotations of afforestation on deep peat could provide net carbon benefits. The soil carbon content of such sites was estimated to fall after 3 rotations. Such model outputs are encouraging but require validation data from Scottish afforested peatlands at present. Other under-reported evidence includes the emissions from drainage grips on afforested peatlands, which can be substantial sources of CO_2 and CH_4 emissions (e.g Minkinnen and Laine, 2006).

Peatland conversion to cultivated land, peat harvesting and peat erosion carry the highest emission factors. The literature values almost universally point to large net losses in such areas. It must be pointed out, however, that the majority of the studies from which the data were derived only monitored soil respiration and did not take into account any photosynthetic uptake. As most of such sites are predominantly bare, however, we assumed photosynthetic uptake to be minimal. Depending on the fertility of the site and any additional fertilisation practices, substantial methane or nitrous oxide emissions may also be observed on cultivated peatlands.

Finally, peatland restoration can achieve very high carbon benefits in the initial years post restoration which then 'tail off' as the restored peatland reaches a more stable phase. This is thought to be due to high inputs from increased primary production in the years after restoration efforts as vegetation re-establishment is often very rapid and strong growth characterises the early phases. The net decomposition pathways in these early years are still running at a lower rate than in a near-natural system as the microbial community is not yet adapted to the increased carbon inputs below-ground. This 'lag effect' may be the cause of the, often high, net carbon benefits reported from restoration sites in the early phases after restoration efforts have been completed. In some sense, such early high carbon benefits may be viewed as similar to the high accumulation rates observed in young mires (e.g. Clymo et al., 1998). The most complete data on the GHG benefits of peatland restoration are the reports by Bortoluzzi et al. (2006), where the data included in Table 1 both CO_2 and CH_4 terms and suggest generally strong carbon benefits even when methane emissions are taken into account.

The net emission factor (EF) in any restoration scenario is the difference between the initial and final EFs, e.g. if a loss of 3.5 t $CO_2e\ ha^{-1}\ yr^{-1}$ can be restored to give a sequestration of 1 t $CO_2e\ ha^{-1}\ yr^{-1}$, the net benefit is 4.5 t $CO_2e\ ha^{-1}\ yr^{-1}$. As expressed above, all emission factors carry with them a very substantial uncertainty, which is primarily caused by large, more than 100%, interannual variability in carbon fluxes. Even in the case of peatlands in relatively good condition, this variability can take the form of an observable carbon sink in one year followed by function as a source the following year. In the absence of sufficient long-term experimental data that can set limits on the interannual variability in peatlands of less than good condition, it is likely that similar uncertainty applies.

Area with potential for restoration

The condition of the Scottish peatland resource is not currently known at national scale. Much of the resource is not in pristine condition, with some areas in exceptionally poor condition due to excessive drainage, overgrazing and/or peat harvesting. Joosten and Clark (2002) put UK peatland area at a conservative 1750 kha but mire at 100 kha, i.e. only 6% is actively peat forming. Since nearly 80% of the resource is in Scotland, the Scottish figure would be close to 6% also. We estimate that only ca. 30% of the Scottish blanket bog area is currently in good, near-natural, condition based on mapping of erosion, forestry conversion, and peat cutting plus a conservative estimate of the existing grazing, burning and windfarm impacts (Artz et al., 2013a). The situation is much worse for raised bog, where estimates suggest that only ca. 2.7 kha (< 10 %) are in near-natural condition (Lindsay & Immirzi 1996; Artz et al., 2013b).

A conservative estimate puts the figure for erosion-impacted blanket bogs in Scotland at a minimum of 157 kha to 390 kha for major erosion features (based on the 1:250,000 Soils Map deep peat soils and LCS88 definition of blanket bog erosion features, respectively, Artz et al., 2013a). A maximum of 706 kha for all LCS88 blanket bog areas are affected by a certain degree of erosion, depending upon the severity of erosion and the percentage of area affected.

Historic or current domestic and industrial peat extraction affects ca. 56.8 kha of blanket bog (mostly domestic extraction) and between 2.9 to 4.1 kha of raised bog (Artz et al., 2013a,b), some of which overlap in area with erosion and other issues. Substantial carbon savings may be achieved in such areas if further losses through erosion can be controlled; however, the extent and severity of erosion and peat harvesting differ, making it difficult to estimate the effect on the condition of such peatlands.

Full conversion to agricultural land is relatively low in occurrence in Scotland with the exception of some former raised bog margins. Excessive grazing, in isolation or combined with historic drainage for agricultural improvement, and drainage in preparation for forestry, are a common feature of many blanket and raised bog areas but reliable figures for the areal extent do not exist (Artz et al., 2013a,b) as much of the early (1940-1980) agricultural grant payments for drainage and payment schemes for grazing did not require monitoring of the magnitude of their impacts. Emissions data from conversion to agriculture and drainage practice, with or without excessive grazing, suggest that substantial carbon savings are possible from such peatlands, but a national assessment of peatland condition would be necessary to enable identification of specific areas suitable for restoration. The figures for afforestation alone (next bullet) suggest that ca. 190 kha of UK deep peatlands were prepared for conifer planting through ploughing and drainage (Morison *et al.*, 2010), suggesting a very large figure for all land use conversions involving drainage. However, the impact of drainage schemes applied before the 1980's, as there are no national data that have monitored if such drains are still functional or have led to other drainage factors such a peat pipe or gully formation.

While the current guidance is avoidance of forestry plantations on peat deeper than 0.5 m, ca. 145 kha of deep peat (soils >0.5 m of peat) have already been converted to commercial conifer plantations and an additional 0.4 kha are under broadleaf woodland cover (Morison *et al.*, 2010). As the carbon savings from especially conifer plantations are predominantly held in the tree biomass, restoration to peatland habitat may hold persuasive potential for long-term carbon savings. Indeed, restoration of formerly afforested peatlands is already common practice or in discussion on many of the Scottish peatlands areas that hold particularly high carbon stocks (e.g. the Flow Country blanket bogs, and the Flanders Moss and Solway Moss North raised bog complexes). However, as previously stated, the carbon benefits of restoration versus another forestry rotation, and indeed, whether restoration is realistically achievable, cannot be ascertained at present for the afforested area in Scotland. This discussion also needs to take into account the other ecosystem services that peatland or plantation forests provide, but this was outside the scope of this report.

Abatement potential over time

There are no published data on restoration projects that are directly comparable to Scottish peatlands. Very few of the studies published from projects in other countries have followed the carbon balance of a restoration project over a long period, and there is no single published study on carbon balances that has followed a restoration project from pre-restoration measures through to the resulting situation after a few years to decades, hence predictions for abatement potential from restoration measures can only carry limited confidence. Current proxy development, such as using vegetation composition as an indicator of the site C emissions (e.g. Couwenberg et al., 2011), show much promise but require validation in a Scottish context.

Various researchers have published data on emissions savings from restored peatlands that have ranged from a substantial carbon emissions benefit as quickly as two years post restoration measures (Waddington et al. 2010) due to substantial abatement of net decomposition by revegetation or the net loss of DOC and POC, to only modest results after 50 years (Yli-Petäys et al. 2007) in the case of a severely damaged, mechanically extracted peatland.

In the longer timescale, actual C gains can be expected if restoration achieves a return to a functional peatland. Lindsay (2010) used a simple peat accumulation model based on decomposition rates to suggest a likely timeframe of 42 years before peatland restoration achieves net C gain. Hence it is likely that the development of the abatement potential over time from any restoration project is heavily dependent on the starting condition, for example, carbon savings from a severely drained peatland may take longer to materialise than from a less affected peatland.

All restoration targeted at reducing emissions from damaged peatlands will produce carbon savings. The time required to achieve significant emissions reduction will vary from a few years (for less severely damaged peatlands) to more than a decade (for more heavily damaged peatlands). However, in the latter case, the long-term emissions savings from such sites may be worth the longer wait for a return on the higher financial investment in restoration management, as such sites are at the higher end of the scale of current emissions (see Figure 1 below).

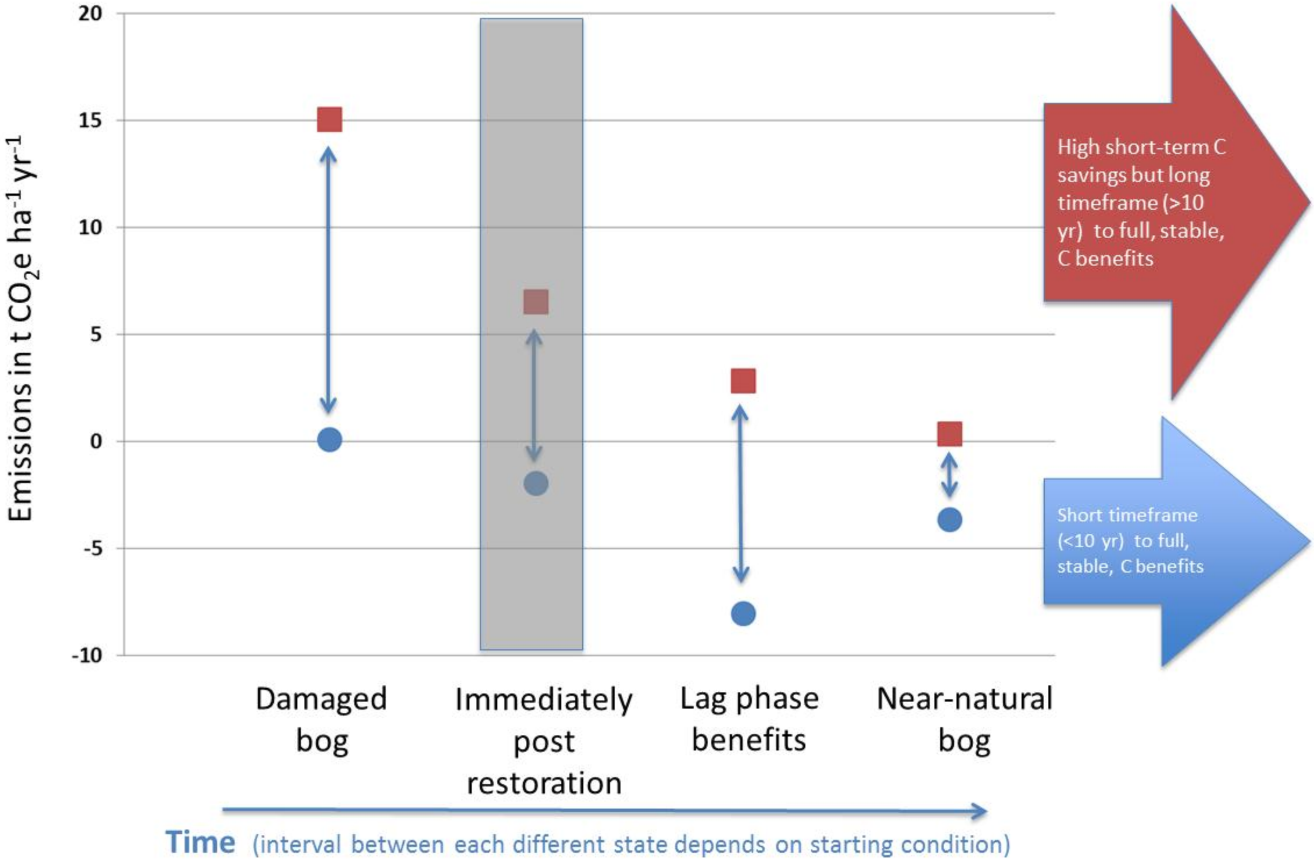

Figure 1: Likely carbon emissions savings between the worst case (red) and best case (blue) extremes of reported literature values for damaged bogs (left) and the trajectory after restoration to the target of creation of stable, near-natural, peatland conditions. High initial carbon savings can be achieved from heavily damaged bogs by limiting erosion and decomposition losses, but full benefits to re-create near-natural bog conditions may take decades. Conversely, lower initial savings may be achieved from restoration of less damaged sites by reversing decomposition losses and enhancing net primary productivity through water-table management, but stable, near-natural, conditions will be achieved in a shorter timeframe, hence creating stable, long-term C benefits much quicker. Data on sites immediately post restoration are highly speculative (grey).

Table 1. Emissions factors (negative values show net uptake, positive values show net loss)

Land use	Likely[$] current average emissions factor (EF) range ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	Literature EF range ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	References *(abbreviation indicates completeness of study: M-modelled, F-full balance, C-CO_2 only, G-GHG only, P-partially modelled, L-long term average)[#.] References in italics indicate studies where a range of literature values has been presented and the full range has been included in this Table.*
Near-natural	-3.7 to +0.3	-4.1 to +0.3	(Gorham 1991[M]; Cannell *et al.* 1999[M]; Saarnio et al., 2007; Charman 2002[L]; Turunen 2003[L]; Billett *et al.* 2004[P]; Worrall *et al.* 2009[F]; Dinsmore *et al.* 2010[F]; Drewer *et al.*, 2010[F]; Evans & Lindsay 2010[P]; Koehler *et al.* 2011[F]; Worrall *et al.* 2011[L])
Bare peat (eroded or harvested[1])	0 to +5.5	0 to +9	(*IPCC 2006*; Bortoluzzi *et al.* 2006[G]; Waddington *et al.* 2010[P]; Evans & Lindsay 2010; *Couwenberg 2011; Couwenberg et al. 2011*; Worrall *et al.* 2011[M])
Afforested[2]	-9 to 4.8 (excluding extremely young and old stands)	-12.5 to 14.7 (stand age dependent)	(*IPCC 2006; Couwenberg 2011*; Lohila *et al.* 2011[G]; Hargreaves et al., 2003[C/P], Minkkinen et al., 1999[P]; Ojanen et al., 2013[M], Lohila et al., 2007[c])
Drained (for forestry or grazing improvements[3])	-0.05 to +5.5	-0.3 to +20	(Rowson *et al.* 2010[F/P]; Lohila *et al.* 2011[G]; *Couwenberg et al. 2011;* Hargreaves et al., 2003[C/P]; *Couwenberg 2011*; Worrall *et al.* 2011[M])
Cultivated	+9.2 to +15	+5.5 to +24	(*Couwenberg et al. 2011; Couwenberg 2011; Maljanen et al., 2007*)
Restored[4]	Highly variable, dependent on site history and time since restoration	-8.1 to +2.8	(Byrne *et al.* 2004; Bortoluzzi *et al.* 2006[G]; Yli-Petäys *et al.* 2007[G]; Waddington *et al.* 2010[P]; Samaritani *et al.* 2011[C])

[$] Excludes data from predominantly modelled studies, or those atypical to the UK situation. Data do not take into account the global warming potential of methane, as not every study differentiated methane fluxes within the stated budget. See Table 2 for complete figures from near-natural peatlands.

M-modelled: all components of the reported balance have been derived from a process-based model; F-full balance: all components of the reported balance have been measured directly at the site, although modelling to provide infill for missing data points will have been carried out; C-CO_2 only: Only carbon dioxide budgets have been measured; G-GHG only: Only carbon dioxide and methane budgets have been measured; P-partially modelled: one or more component terms of the budget are derived from models or estimates; L-long term average: the accumulation rate of the peat itself has been measured.

[1] Recently harvested peatlands or fresh erosion features carry the high-end of the range EF values.

[2] Afforested and cropland sites may also have N_2O emissions arising from fertilisation at time of planting. No UK values for this in existence. Negligible on other sites except where high N deposition exists (Drewer *et al.* 2010).

[3] Drained sites showing vegetation conversion to grassland vegetation probably at the higher end.

[4] Probably insufficient data available at present, all from previously bare sites. Very high net sequestration rates may be observed only temporarily as decomposition pathways are not yet adapted to the new C inputs.

Table 2. Summary of carbon balances from ombrotrophic peatlands in the Northern hemisphere

Site	**year**	**g C m^{-2} yr^{-1}**				**Reference**	**In CO_2e (accounting for global warming potential of GHGs)**		
		CO_2	CH_4	DOC	**NECB**		CO_2	CH_4	**net GHG**
Glencar	2003	-66.8	3.8	13.5	**-49.5**	Koehler et al., 2011	-244.7	126.8	**-117**
Glencar	2004	-67.2	3.6	13.1	**-50.5**	Koehler et al., 2011	-246.2	120.2	**-126**
Glencar	2005	-84	4.5	13.9	**-65.6**	Koehler et al., 2011	-307.7	150.2	**-158**
Glencar	2006	-12.5	4.6	16.5	**8.6**	Koehler et al., 2011	-45.8	153.6	**108**
Glencar	2007	-13.5	4.2	11.9	**2.6**	Koehler et al., 2011	-49.4	140.2	**91**
Glencar	2008	-42.7	3.6	15	**-24.1**	Koehler et al., 2011	-156.4	120.2	**-36**
Loch More	1994		5.2			Hargreaves and Fowler, 1998		172.5	
Glencar	2004		4.6			Laine et al., 2007		152.7	
Stordalen	2008/ 2009	-50	2	3.2	**-44.8**	Olefeldt et al., 2012; Olefeldt and Roulet, 2011	-183	67	**-116**
Stordalen	2003	-27	5			Christensen et al., 2007	-98	166.9	**68**
Fajemyr	2006	-78.6				Lund et al., 2007	-288		
Fort Simpson	1995		4.9			Liblik et al, 1997		164.25	
Lena River Delta	2006		5.1			Sachs et al., 2008		170.5	
James Bay	2004		2.8			Pelletier et al., 2007		95	
Minnesota	1991		12.9			Shurpali et al, 1993		431.2	
North Hudson Bay	1990		0.9			Roulet et al., 1994		32.7	
South Hudson Bay	1990		2.1			Roulet et al., 1994		69.7	
Stordalen	2006		18.3			Jackowicz-Korczynski et al., 2010		612.5	
Stordalen	2007		22.1			Jackowicz-Korczynski et al., 2010		737.5	
Mer Bleue	1999	-60	4	12	**-44**	Roulet et al., 2007	-219.8	133.6	**-86.3**
Mer Bleue	2000	-35	4	13	**-18**	Roulet et al., 2007	-128.2	133.6	**5.3**
Mer Bleue	2001	-3	5	11	**13**	Roulet et al., 2007	-11.0	167.0	**156.0**
Mer Bleue	2002	-19	4	17	**2**	Roulet et al., 2007	-69.6	133.6	**64.0**
Mer Bleue	2003	-15	4	18	**7**	Roulet et al., 2007	-55.0	133.6	**78.6**
Mer Bleue	2004	-115	4	20	**-91**	Roulet et al., 2007	-421.4	133.6	**-287.8**
Ellergower Moss	1992		2.3			Clymo et al., 1995		78.8	
Ellergower Moss	1992		6.4			Clymo et al., 1995		214.3	
Bad a Cheo	1992		3.6			Chapman and Thurlow, 1996		121.9	
Stor Amyran	1992	-10.3	4.1	4.2	**-2**	Waddington and Roulet, 2000	-37.7	136.9	**99.2**
Stor Amyran	1993	-3	3.9	6.7	**7.6**	Waddington and Roulet, 2000	-11.0	130.2	**119.2**
Auchencorth Moss	2006	-88.4	0.1			Drewer et al, 2010	-324.0	4.9	**-319.1**
Auchencorth Moss	2007	-136.0	0.3	18.6	**-117.1**	Dinsmore et al., 2010	-498.3	9.9	**-488.5**
Auchencorth Moss	2008	-93.5	0.4	32.2	**-60.9**	Dinsmore et al., 2010	-342.6	12.3	**-330.3**
Auchencorth Moss	1998	-27.8	4.1	26.9	**3.2**	Billett et al., 2004	-342.6	136.9	**-205.7**
Auchencorth Moss	2011		1.7			Drewer et al, unpublished (poster)		56.7	
Whim Moss	2008		5.0			Kivimaki et al., 2012		167.5	
Average flux (+/- SEM) [n]					**-29.9 +/- 9 [n=19 from 5 sites]**		*-194 +/- 30 [n=22 from 6 sites]*	*157 +/- 25 [n=36 from 16 sites]*	*-76 +/- 39 [n= 21 from 5 sites]*

NECB – net ecosystem carbon balance. Negative values indicate net sequestration into the soil.

Further information

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