Supplementary Material

S1. Derivation of pressure-response functions

Pressure-response functions were obtained from the literature, giving preference to relationships derived from meta-analyses of multiple studies, or syntheses of data from multiple sites, where possible. As noted in the main text, this did not take the form of a formalised literature search, systematic review or meta-analysis, due both to time constraints and to the difficulty of extracting data for multiple response functions from multiple primary literature sources in an appropriate and consistent form. As such, we acknowledge an inevitable risk of incompleteness or subjectivity in the approach taken, although we have endeavoured to identify the most complete and representative data sources in each case. Continuous functions were derived if sufficient data were available to support this, and simpler categorical functions if not. If uncertainty ranges were given in the source publications, or could be derived from the data presented, these were included in the response functions produced. Given the varying quality and quantity of data available from which to derive each response function, each was assigned a qualitative reliability score, following the approach used for nitrogen critical load assessment (e.g. Bobbink and Hettelingh, 2011), as follows:

***Table S1**. Criteria used to define reliability levels of pressure-response functions*

Reliability index	**Description**	**Notation**
'Reliable'	Based on a meta-analysis or synthesis of multiple data sources, showing a consistent response	##
'Quite reliable'	Based on a limited number of studies, or where published data show some variability in response	#
'Expert judgement'	Based on one or a few individual studies with limited spatial coverage.	(#)

Where we were unable to identify sufficient evidence of a causative relationship, or where a relationship was suspected but could not be adequately quantified, a response function was not developed.

S1.1 CO_2 and water table

The relationship between net CO_2 flux to or from the peat surface (i.e. net ecosystem exchange, NEE) and water table has been examined in numerous studies, primarily in relation to the artificial drawdown of water table due to drainage. For this study, we took a relationship from a published meta-analysis of net CO_2 flux versus mean annual water table by Couwenberg et al. (2011), which was derived from measurements reported in ten studies of a range of drained temperate peatlands. The empirical relationship obtained by Couwenberg et al. (2011) is linear to a mean annual water table depth of 50 cm, and can be expressed as:

$$CO_2 \text{ flux (t } CO_2 \text{ ha}^{-1} \text{ yr}^{-1}) = 0.752 \times \text{Water table (cm)} - 4.75 \qquad \text{(S1)}$$

Where positive CO_2 flux denoted CO_2 emission, and a negative flux net CO_2 uptake, and water table is expressed as a (positive) distance below the ground surface. The relationship suggests that peatlands act as a modest CO_2 sink when mean water table is within 6.5 cm of the surface, transitioning to an increasing net CO_2 source as water table drops below this threshold. Below a water table depth of 50 cm, Couwenberg et al. (2011) observed limited evidence for further increases in CO_2 emissions, although a subsequent analysis (Couwenberg and Hooijer, 2012) suggests that emissions may continue to increase with greater water table drawdown. In practice, it would be very unusual for water table in blanket bogs to fall below this level, with mean water table in drained sites typically remaining within 20 cm of the surface across most of the bog area (e.g. Holden et al., 2011; Wilson et al., 2011; Cooper et al., in press). In the Durham Carbon Model, Worrall et al. (2009) linked water table in blanket bogs to drain spacing, providing a potential basis for relating CO_2 flux directly to intensity of management. Drain-blocking would be expected to reverse the increase in CO_2 emissions.

S1.2 CO_2 and managed burning

Despite the prevalence of managed burning across large areas of UK blanket bog, reliable quantitative data on the net impact of this activity on carbon balances are sparse. In part, this is because short-term flux measurements are likely to give different results depending on when they take place during the burn/re-growth cycle. For example, Clay et al. (2010) noted observed net CO_2 drawdown in recently burnt plots, offsetting CO_2 losses during burning. As a result, we were reliant on data from a single long-term experimental study by Garnett et al. (2000) in which peat core carbon accumulation rates were measured above a dateable horizon in replicate rotationally burnt and unburnt plots at Moor House, Northern England. The study provided the basis of a simple (burnt/unburnt) categorical function. The CO_2 sequestration rate for unburnt peat was calculated from the total C accumulated in peat (5.4 kg C m^{-2}) above the dated horizon (1945 ±5 years), divided by the number of years between this point and sample collection in 1995. The resulting calculated CO_2 sequestration rate for unburnt blanket bog is -3.81 t CO_2 ha^{-1} yr^{-1} (note that this is consistent with the value that would be obtained from Equation S1 above, for an undrained bog with a mean annual water table depth of 1-2 cm)

Carbon sequestration rate in burnt peat was estimated from the difference in total C accumulation between the burnt and unburnt cores (2.3 kg C m^{-2}), which was assumed to have resulted from the period of the core record during which managed burning took place (32 years). This gave a 74% reduction in C accumulation rate over this period, implying a mean annual CO_2 sequestration under managed burning of -1.09 t CO_2 ha^{-1} yr^{-1}. The error ranges shown are derived from the 95% confidence intervals on the mean total C accumulation estimates reported by Garnett et al. (2000), together with the quoted uncertainty in the age of the dated horizon.

S1.3 CO_2 and N deposition

Studies of boreal and temperate forest and heathland ecosystems show a reasonably consistent enhancement of soil C stocks as a result of N deposition. For example, data presented in de Vries et al. (2009) suggest a below-ground enhancement of approximately 10 to 25 kg C per kg N in forests, and 20 to 35 kg C kg N^{-1} for heathlands. For peatlands, the effects of N deposition appear more modest, and are probably non-linear. At low N deposition levels, some studies have shown that additional N inputs enhance C accumulation rates (Turunen et al., 2004; Lund et al., 2009). From the data presented by Turunen et al. (2004), the enhancement of C accumulation by N deposition appears to be around 8.5 kg C kg N^{-1}, i.e. considerably lower than the heathland and forest values. However as N inputs increase, key peat-forming species (notably *Sphagnum* mosses), which are adapted to low-nutrient conditions, may be outcompeted by vascular plants with a greater nutrient-demand (e.g. Berendse et al., 2001; Limpens et al., 2003; Bubier et al., 2007; Wiederman et al., 2007). This is reflected in a 'critical load' (an empirically-based threshold level of N deposition beyond which ecosystem damage is considered to occur) of 5 to 10 kg N ha^{-1} yr^{-1} for European blanket bogs (Bobbink and Hettelingh, 2011). As well as affecting the species balance, N deposition changes ecosystem function because vascular plant litter is generally more degradable than that from *Sphagnum*, and peat formation is consequently reduced. Lamers et al. (2000) observed that

Sphagnum became N-enriched above 10 kg N ha^{-1} yr^{-1}), but that its capacity to 'filter' (i.e. retain) N failed above 20 kg N ha^{-1} yr^{-1}. Wiedermann et al. (2007) showed that *Sphagnum* cover halved in a bog exposed to a 30 kg N ha^{-1} yr^{-1} experimental treatment for 8 years, while Sheppard et al. (2011) showed evidence that higher levels of N exposure could be directly toxic to *Sphagnum* and other bog species. The resulting exposure of bare peat and/or transition to vascular plant dominance is likely to lead to the cessation of peat accumulation, and ultimately may lead to net CO_2 loss.

Overall, the available literature clearly point to a non-linear response of bog CO_2 sequestration to N deposition. For N deposition in the range 0 to 10 kg N ha^{-1} yr^{-1} we used the equations given by Turunen et al. (2004) for peat core C accumulation rate versus N deposition, given separately for hummocks and hollows as:

Hummocks: C accumulation rate = 4.20 + 8.91(N deposition) + 0.035 (dd) (S2)
Hollows: C accumulation rate = 111.21 + 8.18(N deposition) + 0.020 (dd) (S3)

Where C accumulation rate is expressed in g C m^{-2} yr^{-1}, N deposition in g N g C m^{-2} yr^{-1}, and dd is the mean annual growing degree days above 5 °C. Assuming an equal coverage of hummocks and hollows, and assigning a value of 1400 for dd based on the data presented by Turunen et al. (2004), we calculated that CO_2 sequestration by bogs is likely to increase only marginally with increasing N deposition, from -3.53 t CO_2 ha^{-1} yr^{-1} under pristine conditions to -3.84 t CO_2 ha^{-1} yr^{-1} at 10 kg N ha^{-1} yr^{-1}. Following Lamers et al. (2000), we considered that further stimulation of CO_2 uptake was unlikely beyond this level of deposition, because their data showed that additional N led to increased N content of biomass rather than increased biomass growth, and CO_2 sequestration was therefore assumed to remain constant. Above 20 kg N ha^{-1} yr^{-1}, the data of Lamers et al. (2000), Bubier et al. (2004) and Sheppard et al. (2011) all indicate detrimental impacts on *Sphagnum* growth and hence likely negative effects on C accumulation, leading initially to reduced CO_2 sequestration and ultimately to CO_2 emission. However, we were unable to derive a quantitative response function for this range, and therefore included only an indicative response function, with CO_2 sequestration declining linearly beyond a deposition level 20 kg N ha^{-1} yr^{-1}, and the peatland ceasing to act as a CO_2 sink at 30 kg N ha^{-1} yr^{-1}.

S1.4 CO_2 and S deposition

We were not able to define a quantitative pressure-response relationship between S deposition and CO_2 flux for blanket bogs. However, data from other ecosystems, such as forests (e.g. Oulehle et al., 2011) indicate that soil acidification due to S deposition can have a strongly suppressive effect on organic matter decomposition, leading to enhanced C accumulation by the ecosystem. Very high rates of S deposition, however, were a major cause of *Sphagnum* dieback in the Southern Pennines during the 20th century (Tallis, 1983; Ferguson and Lee, 1983), leading to severe ecosystem degradation including C loss and the onset of peat erosion. Thus, conceptually the relationship between S deposition and CO_2 flux may show a similar threshold-type reversal to that presented above for N deposition.

S1.5. CH_4 and water table

Emissions of CH_4 from peatlands are strongly related to water table, with maximum emissions when water table is close to the surface. We evaluated two published meta-analyses describing this relationship, by Levy et al. (2012) and Couwenberg et al. (2011). Levy et al. (2012) collated a large dataset of UK CH_4 flux data, mostly from blanket bogs, and found an approximately linear relationship with mean water table down to around 20 cm depth, below which CH_4 was near-zero. Levy et al. (2012) estimated that each 1 cm increase in water table was associated with an emissions increase of 0.4 g CH_4 m^{-2} yr^{-1}. If emissions are considered to be zero below 20 cm water table depth, the resulting equation is:

$$CH_4\ (\text{kg } CH_4\ ha^{-1}\ yr^{-1}) = 80 - 4.0 \times \text{Water Table (cm)} \qquad (S4)$$

Couwenberg et al. (2011) collated data from 16 studies on peatlands in continental Europe, and observed a similar relationship between CH_4 emissions and water table for sites at which aerenchymatous plant species (vascular plants with gas-transporting tissues that act as 'chimneys' for CH_4 emission from the water table to the atmosphere) were present. The resulting equation is:

$$CH_4\ (\text{kg } CH_4\ ha^{-1}\ yr^{-1}) = 334 - 16.7 \times \text{Water Table (cm)} \qquad (S5)$$

The two sets of observations indicate that CH_4 emissions fall to zero at around 20 cm mean water table depth, but for shallower water tables Equation S5 predicts four times higher emissions than Equation S4.

S1.6. CH_4 and S deposition

The relationship obtained by Gauci et al. (2004) was used to define the response of CH_4 emissions to S deposition. Gauci et al. synthesised data from experimental sulphate additions to bogs in the UK, Scandinavia and the United States, and showed a consistent pattern of suppression of CH_4 emissions in response to sulphate, which results from the competition between sulphate reducing bacteria and methanogens for labile organic matter substrates. The suppressive effect is strongest at low S deposition levels, resulting in a non-linear empirical relationship of the form:

$$CH_4 \text{ suppression} = (38.6 \times \text{S deposition})/(\text{S deposition} + 8.71) \qquad (S6)$$

Where CH_4 suppression is expressed as a percentage reduction in emissions relative to the emissions that would occur in the absence of S deposition, which is expressed in kg S ha^{-1} yr^{-1}.

S1.7. CH_4 and managed burning

The direct effects of burning on CH_4 emission appear modest. A small (10-20%) increase in emissions is possible immediately after burning as water tables rise in the absence of vegetation cover (Ward et al., 2007; Clay et al., 2010), but as this effect is transient, the overall change in CH_4 emissions through the burn-regrowth cycle is likely to be negligible. A more important effect of burning may be via the long-term changes in vegetation that are induced by regular burning (e.g. Holden et al., 2012). Since managed burning is designed to increase the dominance of *Calluna vulgaris*, a corresponding reduction in aerenchymatous sedges such as *Eriophorum* species would potentially result in lower CH_4 emissions from rotationally burnt areas. A pressure-response function was not developed for this effect, although the use of vegetation-based proxies for prediction of CH_4 emissions from peatlands (e.g. Couwenberg et al., 2011; Dias et al., 2010; Gray et al., 2013) suggests that a response function based on vegetation cover as an intermediate condition variable (reflecting the effects of burning as well as other environmental drivers) might be developed in future.

S1.8 CH_4 and N deposition

As for burning, the effects of N deposition on CH_4 may be associated with long-term plant species change. Increased nutrient loadings to bogs are known to trigger a shift in species dominance from *Sphagnum* and other bryophyte species towards vascular species, and some vascular species are associated with higher CH_4 emissions (see Section S1.3 and S1.5). However species changes due to N deposition are likely to occur very gradually over long time periods, and are very hard to predict. As for burning effects, we did not derive a direct pressure-response function for this driver, but note that a response function based on vegetation condition might be appropriate.

S1.9. DOC and drainage

Empirical data describing the effects of peatland drainage on DOC losses were recently collated for the IPCC Wetland Supplement (IPCC, 2013). In all, data from twelve sites (including data from one study published since the IPCC report) were collated, each of which recorded DOC concentrations in either peat porewater or drainage water from comparable drained and undrained sites. Note that data from tropical peatlands included in the IPCC report have been omitted here, since these sites are not comparable to blanket bogs. All boreal and temperate peatland studies have been included, however, since only one study took place on a blanket bog. Overall, the collated data (Table S2) show a fairly consistent tendency towards higher DOC concentrations in drained peatlands, with a mean concentration increase of 60% (note that the single blanket bog study gave a value close to this mean). The limited number of studies, variability of sites and methods, and lack of detailed water table data meant that we were not able to derive a continuous response function from these data, and a simple (drained/undrained) categorical approach was therefore taken.

***Table S2**. Published studies containing data on DOC response to drainage*

Study	Country	Peat type	DOC concentrations (mg/l)		Increase (%)
			Undrained	Drained	
Glatzel et al. (2003)	Canada	Boreal bog	60	110	83%
Strack et al. (2008)	Canada	Boreal fen	16	24	53%
Kane et al. (2010)	USA	Boreal fen	56	72	29%
Heikkinen (1990)	Finland	Boreal fen	17	20	15%
Moore et al. (2007)	New Zealand	Temperate bog	70	108	54%
Urbanova et al. (2011)	Czech Rep.	Temperate bog	36	54	51%
		Temperate fen	17	38	118%
Banas and Gos (2004)	Poland	Temperate bog (*Sphagnum*)	31	65	113%
		Temperate bog (Pine)	54	75	39%
		Temperate bog (Birch)	58	73	26%
Frank et al. (2014)	Germany	Temperate bog	49	89	84%
Wallage et al. (2006)	UK	Blanket bog	28	43	55%

To convert observed concentration increases into fluxes, we followed the IPCC (2013) approach and firstly estimated the DOC flux for undrained blanket bogs based on measurements from five near-natural British and Irish blanket bogs (Dawson et al., 2004; Dinsmore et al., 2011; Billett et al., 2010; Koehler et al., 2009, 2011), giving a mean DOC flux of 22 g C m^{-2} yr^{-1}. Applying the 60% increase in DOC concentrations across all drained peatland sites derived from the studies in Table S2, and assuming no concurrent change in annual water flux (which is considered unlikely in high-rainfall blanket bogs), the estimated DOC flux from a drained blanket bog is 35 g C m^{-2} yr^{-1}. Errors were calculated as 95% confidence intervals derived from the standard error of the mean reported natural peatland flux and mean reported DOC concentration increase following drainage.

Data from a smaller set of peat re-wetting studies (Wallage et al., 2006; Gibson et al., 2009; Armstrong et al., 2010, Frank et al., 2014) show some variability, but overall suggest that ditch-blocking has the opposite effect on DOC compared to drainage, and that the average magnitude of DOC reduction following re-wetting is approximately equivalent to the magnitude of DOC increase after drainage. As such, the response function produced appears to be reversible, although this does not rule out transient effects in the period immediately after re-wetting.

S1.10. DOC and burning

Evidence for the effects of managed burning on DOC loss is limited, and some studies have produced conflicting results, as discussed below. With insufficient data from which to calculate a continuous response function, we used a simple categorical approach as for drainage. For unburnt catchments, we assigned the same DOC flux as for undrained catchments (see preceding section). For burnt catchments, we took data from a study of three Pennine blanket bog catchments subject to managed burning by Yallop et al. (2010). Comparing data from six different years, they observed a positive linear relationship between calculated DOC flux and the area of recently burnt blanket bog, as derived from aerial photographs taken in the same year (see Figure 4a of Yallop et al., 2010). This relationship suggests that the DOC flux from an area of peat that had been entirely burnt would be approximately 54 g C m^{-2} yr^{-1}. Note that we could not derive a confidence interval around this value from the available data. It is also worth noting that in reality it would be highly unusual for an entire blanket bog area to be burnt, unless due to wildfire, because grouse moor management involves the rotational burning of smaller patches, so in practice (as observed by Yallop et al., 2010) the proportion of recently burnt peat is unlikely to exceed around one third of the total area and effects on DOC are correspondingly smaller. This was taken into account in model predictions by assigning the actual area of recent burn within a given area (see Section S2).

Finally, it is important to state that the magnitude of DOC burning responses inferred from monitoring data by Yallop et al. (2010) have been challenged (e.g. Chapman et al., 2010; Holden et al., 2012), and that similar responses were not observed in plot-scale burning experiments (Ward et al., 2007; Worrall et al., 2007). Holden et al. (2012) suggest that higher DOC losses from burnt catchments could be associated with the resulting increase in heather cover, rather than burning *per se*, while Clay et al. (2012) observed an increase in water colour but not DOC concentration in the four years following burning, suggesting a qualitative rather than quantitative change in DOC loss. In summary, there is significant uncertainty for this response function, and it was assigned the lowest reliability score.

S1.11 DOC and S deposition

Changes in S deposition have now been identified as a key control on DOC leaching in many ecosystems, via the effects of sulphate on acidity, and subsequently on the solubility of organic matter (e.g. Evans et al., 2012). The most comprehensive analysis of this effect to date was undertaken by Monteith et al. (2007), who analysed long-term surface water data from over 500 sites in Europe and North America, from which a simple empirical equation was derived relating the observed change in DOC leaching due to S deposition (%ΔDOC(anthropogenic)) to the observed change in surface water non-marine sulphate concentration (ΔxSO_4^{2-}, in μeq l^{-1})

$$\%\Delta DOC(\text{anthropogenic}) = -0.557\ \Delta xSO_4^{2-} \qquad (S7)$$

This equation was used to define a continuous pressure-response function for the effects of S deposition on DOC leaching, relative to the natural blanket bog DOC flux of 22 g C m^{-2} yr^{-1} estimated above.

S1.12 DOC and N deposition

The extent of DOC response to N deposition is debatable (e.g. Evans et al., 2008) and, as for CH_4 emissions, any response may be associated with long-term productivity and plant species changes due to eutrophication (Rowe et al., 2014), rather than to direct short-term effects.

S1.13. POC response function

Significant POC losses occur where anthropogenic pressures (or some natural processes) lead to exposure of the peat surface or increased overland flow. Historically, severe peat erosion in the Southern Pennine area of Northern England is thought to have been triggered by a combination of S deposition, wildfire and over-grazing (e.g. Tallis, 1987, 1997). Managed burning is likely to differ in impact from wildfire, as plant roots should not be damaged, but data are sparse; Clay et al. (2013) observed a peak POC loss of 32 g Cm^{-2} yr^{-1} in the year after burning, over ten times higher than controls, but a return towards unburnt flux levels towards the end of the management cycle. Drainage also enhances POC loss, and substantial POC flux reductions have been recorded in drain-blocked catchments (Holden et al., 2007b).

While POC responses to these individual pressures are difficult to characterise, and validated process models of POC flux from peatlands are not yet available, there is scope to develop condition-response functions using bare peat area as a predictor for POC flux which reflects the impact of multiple pressures. We developed a response function by combining published data from a number of sources, as follows:

i) Average surface recession rates for exposed peat on blanket bog were calculated by Evans and Warburton (2007), based on a set of 17 erosion pin studies, to be 22.5 mm yr^{-1}, with a 95% confidence interval of 8.0 mm yr^{-1}. A single figure for all exposed blanket peat appears reasonable given the relatively narrow range of climatic conditions under which blanket peat formation occurs (Gallego-Sala and Prentice, 2012).
ii) The proportion of this total peat loss exported as waterborne POC was estimated, based on Evans and Warbuton (2007), to be 38% (the remaining loss occurs via wind erosion and oxidation).
iii) Combining this figure with a typical dry peat bulk density of 0.1 g cm^{-3} (Evans and Warburton, 2007) and a C content of 48% (Pawson, 2007) gives a POC loss flux from fully exposed peat of 4.1 t C ha^{-1} yr^{-1}.

On the basis of this calculation, we derived a linear response function for POC loss, assuming zero loss from peat with no bare area, and a loss of 4.1 t C ha^{-1} yr^{-1} (95% confidence interval 2.65 to 5.56 t C ha^{-1} yr^{-1}) from fully exposed peat.

S1.14 Overland flow response function

We based our assessment for overland flow on the work of Holden *et al.* (2008) who showed, using plot-scale measurements at 256 locations within a UK blanket bog catchment, that vegetation type strongly influences the velocity of overland flow across blanket bogs. *Sphagnum* was found to have a retarding influence relative to areas of sedge (*Eriophorum spp.*), with bare peat having the highest velocities. Holden et al. used their results, obtained from measurements on different slopes, water depths and vegetation, to develop an equation to predict overland flow velocity if the topography, vegetation cover and rainfall are known. This simple model has the potential to be applied more broadly as part of an ecosystem service assessment. However for the purposes of this study we took a simpler categorical approach, assigning mean flow velocities for different discharge rates based on the data presented in Table 1 of Holden et al. (2008). Velocity data are provided for *Sphagnum*, mixed *Sphagnum-Eriophorum*, *Eriophorum* and bare peat. Previous work (Bonn et al., 2010) has suggested that, conceptually, *Calluna*-dominated areas are likely to have flow velocities between that of *Eriophorum* and bare peat.

S2. Application of DOC response functions for large-scale spatial modelling

The following section describes the calculations steps carried out on a gridded basis in order to simulate the effects of changes in S deposition, drainage and burning relative to pre-industrial levels on the leaching of DOC from blanket bogs, which was undertaken for the case study area of Northern England. The overall procedure is described in Section 2.2 of the main text, together with the scenarios used, data sources and basis for fixed parameter values used. For each cell, a ratio of modelled to reference (i.e. undisturbed) DOC leaching flux was calculated for each of the three drivers, as follows:

S2.1 Effects of S deposition

For scenario year t, the following equations were applied sequentially to each grid cell:

i) $[xSO4]_t = Sdep_t \times 100/32 \times (1\text{-}fSret) / Q$

ii) $\Delta[xSO4]_t = [xSO4]_t - [xSO4]_0$

iii) $\%\Delta DOC_{s,t} = -0.477 \times \Delta[xSO4]_t$

iv) $RDOC_{s,t} = (100 + \%\Delta DOC_{s,t})/100$

Where:

Sdep is the non-marine sulphur deposition for the scanario year, in kg S ha^{-1} yr^{-1}

fSret is the fraction of deposited sulphur retained by the peat rather than leached as sulphate.

Q is the annual water flux from the site in m yr^{-1}

$[xSO4]_t$ is the calculated non-marine sulphate concentration in scenario year t, in μeq l^{-1}

$[xSO4]_0$ is the calculated non-marine sulphur concentration under pre-industrial conditions, in μeq l^{-1}

$\Delta[xSO4]_t$ is the difference in non-marine sulphur concentration between the scenario year pre-industrial conditions, in μeq l^{-1}

$\%\Delta DOC_{s,t}$ is the percentage change in DOC (concentration or flux) due to sulphur deposition in scenario year t, relative to the pre-industrial reference level.

$RDOC_{s,t}$ is the ratio of DOC (concentration or flux) due to sulphur deposition in scenario year t, relative to the pre-industrial reference level.

S2.2 Effects of managed burning

The following equations were used to predict the effect of managed burning on DOC loss:

i) $Area_{Class1} = Area_{Burn} \times Frac_{Class1}$

ii) $RDOC_{b,t} = 1 + \{Area_{Class1} \times (RDOC_{Class1} - 1) / 100\}$

Where:

$Area_{Burn}$ is the area of the grid cell subject to burn management, expressed as a percentage of the total area.

$Area_{Class1}$ is the area of the grid cell subject to recent 'Class 1' burn, as defined by Yallop et al. (2008).

$Frac_{Class1}$ is the proportion of the total area subject to burn management classed as Class 1 recent burn.

$RDOC_{b,t}$ is the ratio of DOC loss (concentration or flux) due to burning in scenario year t relative to the unburnt pre-industrial reference level.

$RDOC_{Class1}$ is the ratio of DOC loss (concentration or flux) from Class 1 burnt areas relative to unburnt areas, taken from the response function.

S2.3 Effects of drainage

The following equation was used to predict the effects of drainage on DOC loss:

i) $RDOC_{d,t} = 1 + \{Area_{drained} \times (RDOC_{drained} - 1)\}$

Where:

$Area_{Drained}$ is the percentage of the grid cell which has been drained

$RDOC_{b,t}$ is the ratio of DOC loss (concentration or flux) due to drainage in scenario year t relative to the undrained pre-industrial reference level.

$RDOC_{drained}$ is the ratio of DOC loss from drained area relative to undrained areas, taken from the response function

S2.4 Combined effects of all three drivers on DOC concentrations

The three ratios were combined in order to give an overall predicted effect on DOC leaching, assuming a multiplicative effect. The

i) $RDOC_{combined,t} = RDOC_{s,t} \times RDOC_{b,t} \times RDOC_{d,t}$

ii) $FDOC_t = FDOC_0 \times RDOC_{combined,t}$

iii) $[DOC]_t = FDOC_t / Q$

Where:

RDOCcombined,$_t$ is the overall ratio of DOC flux in scenario year t, relative to that from an undisturbed peatland.

FDOC$_t$ is the DOC flux in scenario year t, in g C m^{-2} yr^{-1}

FDOC$_0$ is the DOC flux from an undisturbed peatland (see Section S1.9), in g C m^{-2} yr^{-1}

[DOC]$_t$ is the DOC concentration in runoff from the peatland in scenario year t, in mg l^{-1}

Supplementary References

(References already included in the main text are omitted)

Armstrong, A., Holden, J., Kay, P., McDonald, A.T., Gledhill, S., Foulger, M., Walker, A., 2010. The impact of peatland drain-blocking on dissolved organic carbon loss and discolouration of water; results from a national survey. Journal of Hydrology, 381, 112-120.

Banaś, K., Gos, K., 2004. Effect of peat-bog reclamation on the physico-chemical characteristics of ground water in peat. Polish Journal of Ecology, 52, 69-74.

Billett M.F., Charman, D.J., Clark, J.M., Evans, C.D., Evans, M.G., Ostle, N.J., Worrall, F., Burden, A., Dinsmore, K.J., Jones, T., McNamara, N.P., Parry, L., Rowson, J.G., Rose, R., 2010. Carbon balance of UK peatlands: current state of knowledge and future research challenges. Climate Research, 45, 13-29.

Berendse, F., Van Breemen, N., Rydin, H., Buttler, A., Heijmans, M., Hoosbeek, M.R., Lee, J.A., Mitchell, E., Saarinen, T., Vasander, H. and Wallen, B., 2001. Raised atmospheric CO_2 levels and increased N deposition cause shifts in plant species composition and production in *Sphagnum* bogs. Global Change Biology 7, 591-598.

Bobbinck, R., Hettelingh, J-P. (Eds), 2011. Review and revision of empirical critical loads and dose-response relationships. Proceedings of an expert workshop, Noordwijkerhout, 23-25 June 2010. RIVM report: 680359002, Coordination Centre for Effects, Bilthoeven.

Bubier, J.L., Moore, T.R., Bledzki, L.A., 2007. Effects of nutrient addition on vegetation and carbon cycling in an ombrotrophic bog. Global Change Biology, 13, 1168–1186.

Chapman, P.J., McDonald, A.T., Tyson, R., Palmer, S.M., Mitchell, G., Irvine, B., 2010. Changes in water colour between 1986 and 2006 in the headwaters of the River Nidd, Yorkshire, UK. Biogeochemistry 101, 281-294.

Clay, G.D., Worrall, F., Aebischer, N.J., 2012. Does prescribed burning on peat soils influence DOC concentrations in soil and runoff waters? Results from a 10 year chronosequence. Journal of Hydrology 448, 139-148.

Couwenberg, J., Hooijer, A., 2013. Towards robust subsidence-based soil emission factors for peat soils in Southeast Asia, with special reference to oil palm plantations. Mires and Peat, 12, Article 01.

Dawson, J.J.C., Billett, M.F., Hope, D., Palmer, S.M., Deacon, C.M., 2004. Sources and sinks of aquatic carbon in a peatland stream continuum. Biogeochemistry 70, 71–92.

Dias, A.T.C., Hoorens, B., Van Logtestijn, R.S.P., Vermaat, J.E., Aerts, R. (2010). Plant species composition can be used as a proxy to predict methane emissions in peatland ecosystems after land-use changes. Ecosystems, 13, 526-538.

Dinsmore, K.J., Billett, M.F., Skiba, U.M., Rees, R.M., Drewer, J., Helfter, C., 2010. Role of the aquatic pathway in the carbon and greenhouse gas budgets of a peatland catchment. Global Change Biology, 16, 2750-2762.

Evans, C.D., Goodale, C.L., Caporn, S.J.M., Dise, N.B., Emmett, B.A., Fernandez, I.J., Field, C.D., Findlay, S.E.G., Lovett, G.M., Meesenburg, H., Moldan, F., Sheppard, L.J., 2008. Does elevated nitrogen deposition or ecosystem recovery from acidification drive increased dissolved organic carbon loss from upland soil? A review of evidence from field nitrogen additions experiments. Biogeochemistry 91, 13-35.

Evans, C.D., Jones, T.G., Burden, A., Ostle, N., Zieliński, P., Cooper, M.D.A., Peacock, M., Clark, J.M., Oulehle, F., Cooper, D., Freeman, C., 2012. Acidity controls on dissolved organic carbon mobility in organic soils. Global Change Biology, 18, 3317–3331.

Frank, S., Tiemeyer, B., Gelbrecht, J., Freibauer, A., 2014. High soil solution carbon and nitrogen concentrations in a drained Atlantic bog are reduced to natural levels by 10 years of rewetting. Biogeosciences, 11, 2309-2324.

Gibson, H.S., Worrall, F., Burt, T.P., Adamson, J.K., 2009. DOC budgets of drained peat catchments: implications for DOC production in peat soils. Hydrological Processes, 23, 1901-1911.

Glatzel, S., Kalbitz, K., Dalva, M. Moore, T., 2003. Dissolved organic matter properties and their relationship to carbon dioxide efflux from restored peat bogs. Geoderma 113, 397-411.

Heikkinen, K., 1990. Transport of organic and inorganic matter in river, brook and peat mining water in the drainage basin of the River Kiiminkijoki. Aqua Fennica, 20, 143-155.

Holden, J., Wallage, Z.E., Lane, S.N., McDonald, A.T., 2011. Water table dynamics in undisturbed, drained and restored blanket peat. Journal of Hydrology 402, 103-114.

Kane, E.S., Turetsky, M.R., Harden, J.W., McGuire, A.D., Waddington, J.M., 2010. Seasonal ice and hydrologic controls on dissolved organic carbon and nitrogen concentrations in a boreal-rich fen. Journal of Geophysical Research 115, G04012. doi:10.1029/2010JG001366.

Koehler, A-K., Murphy, K., Kiely, G., Sottocornola, M., 2009. Seasonal variation of DOC concentration and annual loss of DOC from an Atlantic blanket bog in South Western Ireland. Biogeochemistry, 95, 231–242.

Koehler, A-K., Sottocornola, M., Kiely, G., 2011. How strong is the current carbon sequestration of an Atlantic blanket bog? Global Change Biology, 17, 309–319.

Lamers, L.P.M., Bobbink, R., Roelofs, J.G.M., 2000. Natural nitrogen filter fails in polluted raised bogs. Global Change Biology 6, 583-586.

Limpens, J., Berendse, F., 2003. How litter quality affects mass loss and N loss from decomposing Sphagnum, Oikos, 103, 537–547.

Lund, M., Christensen, T.R., Mastepanov, M., Lindroth, A., Ström, L., 2009. Effects of N and P fertilization on the greenhouse gas exchange in two northern peatlands with contrasting N deposition rates. Biogeosciences, 6, 2135-2144.

Moore, T.R. and Clarkson, B.R., 2007. Dissolved organic carbon in New Zealand peatlands. New Zealand J. Marine Freshwater Res. 41: 137-141.Moore, T.R., Matos, L., Roulet, N.T. 2003. Dynamics and chemistry of dissolved organic carbon in Precambrian Shield catchments and an impounded wetland. Canadian Journal of Fisheries and Aquatic Science 60, 612-623.

Oulehle, F., Evans, C.D., Hofmeister, J., Krejci, R., Tahovska, K., Persson, T., Cudlin, P., Hruska, J., 2011. Major changes in forest carbon and nitrogen cycling caused by declining sulphur deposition. Global Change Biology, 17, 3115-3129.

Pawson, R. R. (2007) Assessing the role of particulates in the fluvial organic carbon flux from eroding peatland systems. Unpublished PhD thesis. School of Environment and Development, University of Manchester. 304p.

Rowe, E.C., Tipping, E., Posch, M., Oulehle, F., Cooper, D.M., Jones, T.G., Burden, A., Hall, J., Evans, C.D., 2014. Predicting nitrogen and acidity effects on long-term dynamics of dissolved organic matter. Environmental Pollution, 184, 271-282.

Sheppard, L.J., Leith, I.D., Mizunuma, T., Cape, J.N., Crossley, A., Leeson, S., Sutton, M.A., van Dijk, N., Fowler, D., 2011. Dry deposition of ammonia gas drives species change faster than wet deposition of ammonium ions: evidence from a long-term field manipulation. Global Change Biology, 17, 3589-3607.

Strack, M., Waddington, J.M., Bourbonniere, R.A., Buckton, L., Shaw, K. Whittington, P., Price, J.S., 2008. Effect of water table drawdown on peatland dissolved organic carbon export and dynamics. Hydrological Processes, 22, 3373-3385.

Urbanová, Z., Picek, T, Bárta, J., 2011. Effect of peat re-wetting on carbon and nutrient fluxes, greenhouse gas production and diversity of methanogenic archaeal community. Ecological Engineering, 37, 1017-1026.

Wallage, Z.E., Holden, J., McDonald, A.T., 2006. Drain blocking: An effective treatment for reducing dissolved organic carbon loss and water discolouration in a drained peatland. Science of the Total Environment, 367, 811-821.

Ward, S.E., Bardgett, R.D., McNamara, N.P., Adamson, J.K., Ostle, N.K., 2007. Long-term consequences of grazing and burning on northern peatland carbon dynamics. Ecosystems 10, 1069-1083.

Wiedermann. M.M., Nordin, A., Gunnarsson, U., Nilsson, M.B., and Ericson, L. (2007). Global change shifts vegetation and plant-parasite interactions in a boreal mire. Ecology, 88, 454-464.

Wilson, L., Wilson, J., Holden, J., Johnstone, I., Armstrong, A., Morris, M., 2011. Recovery of water tables in Welsh blanket bog after drain blocking: Discharge rates, time scales and the influence of local conditions. Journal of Hydrology 391, 377-386.