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The impact of sheep grazing on the carbon balance of a peatland

Fred Worrall *, Gareth D. Clay

Department of Earth Sciences, University of Durham, Science Laboratories, South Road, Durham, UK

HIGHLIGHTS

► Vegetative carrying capacity was from 1.7 to 0.2 ewes/ha over an altitudes up to 900 m asl.
► The GHG carrying capacity was up to 46% lower than the vegetative carrying capacity.
► GHG fluxes varied between a net source of 165 and 195 tonnes CO_2 eq/km^2/yr.
► 91% of the fluxes were directly from the sheep and are therefore not unique to a peat soil.
► The study suggests that emission factors for upland sheep have been underestimated.

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ABSTRACT

Estimates of the greenhouse gas (GHG) fluxes resulting from sheep grazing upon upland peat soils have never been fully quantified. Previous studies have been limited to individual flux pathways or to comparing the presence to the absence of sheep grazing. Therefore, this study combines a model of the physical impact of grazing with models of: biomass production; energy usage in sheep; and peat accumulation. These combined modelling approaches enabled this study to consider the indirect and direct impacts of sheep upon the carbon and greenhouse gas balance of a peatland at different grazing intensities as well as the changes between grazing intensities. The study considered four vegetation scenarios (*Calluna* sp., *Molinia* sp.; reseeded grasses, and *Agrostis-Festuca* grassland) and a mixed vegetation scenario based upon the vegetation typical of upland peat ecosystems in northern England. Each scenario was considered for altitudes between 350 and 900 m above sea level and for grazing intensities between 0.1 and 2 ewes/ha. The study can show that the total GHG flux at the vegetative carrying capacity tended to decline with increasing altitude for all vegetation scenarios considered except for *Molinia* sp. The average total GHG flux for all scenarios was 1505 kg CO_2eq/ha/yr/(ewe/ha), and on average 89% of the fluxes were directly from the sheep and not from the soil, and are therefore not unique to a peat soil environment. The study suggests that emission factors for upland sheep have been greatly underestimated. By comparing the total flux due to grazers to the flux to or from the soil that allows the study to define a GHG carry capacity, i.e. the grazing intensity at which the flux due to grazing is equal to the sink represented by the peat soils, this GHG carrying capacity varies between 0.2 and 1.7 ewes/ha with this capacity declining with increasing altitude for all model scenarios.

1. Introduction

The northern peatland carbon store is estimated to be approximately 4.5 Gt C and over the Holocene northern peatlands have accumulated carbon at an average rate of 0.96 Mt C/yr (Gorham, 1991), making this ecosystem not only a substantial store but also a large potential sink of atmospheric carbon. Unlike many areas of the northern hemisphere UK peatlands are intensely managed and have been impacted by a legacy of atmospheric pollution, tourism, overgrazing and wildfire; and not all of these land management systems have been conducive to carbon storage.

Land management practices represent both a threat and an opportunity with respect to the carbon (C), and greenhouse gas (GHG) budgets of peat soils: a threat because the management may damage the peat and cause a decrease in the magnitude of the carbon sink or even convert the peat soil to a net source of carbon. However, land management can also represent an opportunity to improve carbon uptake in these vital terrestrial carbon stores as the management practice can more readily be altered than external drivers such as increases in air temperature. Furthermore, damage to peatlands can be restored (Worrall et al., 2011), and given the pressures upon UK peatlands, there is more scope for management change than perhaps many other peatlands elsewhere in the World. The UK has committed itself to 80% cuts in GHG emissions by 2050 (Climate Change Act 2008 — www.legislation.gov.uk) and so carbon storage in land will be part of that and with respect to land carbon storage, changes

* Corresponding author. Tel./fax: +44 191 334 2295.
E-mail address: Fred.Worrall@durham.ac.uk (F. Worrall).

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in peat management represent the best opportunity for the largest gains.

Sheep grazing is a common use of land on upland peat soils; however, studies of the impact of sheep have been limited in two respects. Firstly, studies have tended to focus on the presence/absence of sheep rather than the number of sheep present (e.g. Worrall et al., 2007a), and secondly, the studies have not considered the C or GHG impacts of grazing sheep. There are plenty of studies of the effect of the presence of sheep-grazing on vegetation, e.g. Ball (1974) and Hope et al. (1996). Increased grazing has been associated with soil erosion in uplands (Evans, 1996) and a range of studies have associated increased sheep numbers with decreasing soil infiltration rates (Evans, 1990; Langland and Bennett, 1973) and even soil structural collapse (Holman et al., 2003). Burt and Gardiner (1984) demonstrated that decreased soil infiltration coupled with loss of vegetation due to grazing can lead to increased surface runoff. Meyles et al. (2006) have argued that more intense grazing causes conditions which promote increased delivery of soil water to rapid flowpaths. However, none of these studies can draw a correlation between the effect and number of sheep, e.g. if sheep numbers went from 1.2 ewes/ha to 1.5 ewes/ha what difference would it make?

The second limitation of the existing grazing studies is the lack of attention to the affects for C and GHG balance of the grazed environments. There is some information on increased soil erosion, and therefore, increased particulate organic carbon (POC) loss (e.g. Mackay and Tallis, 1996), but trigger levels of grazing for this to occur are not known. Worrall et al. (2007a) showed that there was a significant rise in the water table in a peat soil when grazing was present and that dissolved organic carbon (DOC) concentrations significantly decreased. Clay et al. (2009) showed no significant difference in the concentrations of DOC between grazed and ungrazed plots. Ward et al. (2007) have demonstrated significant increases in net ecosystem exchange (NEE) on grazed vs. ungrazed plots which they ascribe to changes in vegetation community. Clay et al. (2010) have compared the carbon flux between grazed and ungrazed sites on peat soils and suggest that although both types of site were net sources of carbon, the net source represented by the grazed site was on average 37 gC m^2/yr, or 25% lower than the net source represented by the ungrazed sites. Nevertheless, none of these studies were relative to grazing intensity.

The impact of grazing upon the carbon balance of peatlands can be considered under several categories:

1. Direct impact of sheep — Sheep convert plant primary productivity into CO_2, CH_4, wool, meat, faeces and urine. Sheep convert some of its consumption to CH_4 via fermentation and CH_4 has a higher GHG warming potential than CO_2 (Houghton et al., 1996). Equally, some of the egesta of sheep will decompose and release nitrous oxides (N_2O) an even more powerful greenhouse gas than CH_4 (Solomon et al., 2007).
2. Physical impact of the sheep — APART from a sheep's consumption and production, the sheep has an impact upon the carbon budget via its physical presence, e.g. trampling. Physical changes to soils from increased trampling or compaction include increased soil bulk density, decreased air permeability, reduced infiltration, and changed bearing capacity (Willatt and Pullar, 1983). However, the literature on the impact of trampling on carbon flows from soils is sparse. Pengthamkeerati et al. (2005) studied a grazed mineral soil and show CO_2 efflux is significantly reduced with increased bulk density and Hynst et al. (2007) observe lower CO_2 emissions on severely grazed (and therefore trampled) plots. Beare et al. (2009) observe increased DOC (and also CO_2 production) on uncompacted soils in comparison to compacted soils. Of the few studies that investigate trampling and carbon on UK peats, Robroek et al. (2010) show that the absence of vegetation from trampled research tracks on a blanket bog led to an increase in runoff events and transport of particulate organic carbon (POC) and that rapid recovery of the vegetation resulted in cessation of these effects.
3. Peat formation foregone — Plant primary productivity will produce litter which is relatively refractory, i.e. less-readily turned over in the environment, compared to dung and urine, and so the sheep is converting C into a form that is more readily decomposed and lost from the environment. So, therefore, grazing not only restricts an environments ability to produce litter but also converts some primary productivity that would be litter into more readily decomposable organic matter and so the capacity of the environment to lay down layers of peat is reduced.
4. Biomass foregone — A further impact of sheep grazing is that the presence of sheep restricts the capacity of the biomass to grow and regenerate and therefore, at steady state, the above ground biomass will be less where there are grazers as compared to when there are none. The amount of aboveground biomass present on a site represents a stock of carbon.

This study proposes that while the C and GHG flows due to the mass balance of sheep production may not presently be known, the modelling of these budgets is tractable given current available information.

The approach taken by this study considers the impact of sheep grazing to be multiple and consists of: physical impact of sheep upon the fluxes from peat soil; direct losses from sheep; carbon accumulation foregone; and biomass foregone (Fig. 1). When considering the GHG impact of sheep grazing the national GHG inventories would consider the impact of sheep grazing in two distinct ways, firstly the impact on the land and soil, and secondly, the emissions from the sheep. The former would be considered as part of land-use; land-use change and forestry (LULUCF — MacCarthy et al., 2010) and the latter as part of agricultural management (MacCarthy et al., 2010). The reason for this division is that sheep can be moved. If this paper were to consider the physical boundary of the study to be the area of upland peat soil within the UK then a simple technique for managing emissions from grazed peatlands would be to remove all sheep to lowland or mineral soils which would reduce emissions from peatlands but would not reduce total emissions. Therefore, in this study we do consider GHG emissions relative to both the boundary of the peat soils and the boundary of the UK, i.e. this study considered losses from the soil and from the soil plus the sheep upon it.

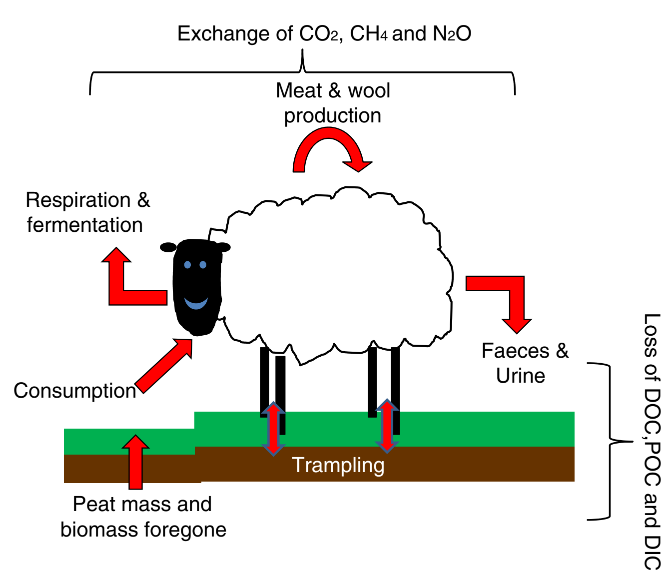

Fig. 1. A conceptual diagram of the fluxes, transfers, flows and processes considered in assessing the greenhouse gas budget of sheep grazing.

2. Approach and methodology

2.1. Physical impact

The physical impact of sheep grazing was considered as a number of impacts. These include the bare soil generated as part of the resting and camping sites and the change in the depth to the water table due to trampling and the loss of biomass due to camping and resting. The former would cause a decline in the primary productivity of a grazed area and the increase in bare soil with increased rest or camping areas would lead to increased POC losses as erosion increases. The change in water table due to grazing would lead to changes in almost all carbon fluxes from a peat except the uptake of CO_2 by primary productivity (P_g) and the POC flux, i.e. it would be expected that fluxes of DOC, net ecosystem respiration of CO_2 (R_{eco}), dissolved CO_2 and CH_4 fluxes would be affected by changes in the depth to the water table.

In order to estimate the physical impact of sheep grazing the Durham Carbon Model was used (Worrall et al., 2009b). The Durham Carbon Model adopts a structural modelling approach, bringing together existing and validated models to calculate as complete a carbon budget as possible for managed peat soil and its catchment. The model considers the following carbon input pathways: CO_2 uptake by primary productivity; and atmospheric inputs as both wet and dry deposition. For carbon release pathways the approach considers both direct release to the atmosphere and fluvial release. For fluvial release the following pathways were considered: DOC, POC, and dissolved CO_2, where each of these species is as defined by Dawson et al. (2002). For direct release of carbon to the atmosphere the approach considers net ecosystem respiration of CO_2; and release of CH_4. Dissolved CH_4 was considered negligible (Hope et al., 2001). The core of the modelling approach was outlined in Worrall et al. (2007b) but with a number of significant improvements described in Worrall et al. (2009b).

The impact of grazing was considered as the difference between the carbon budget of the site at a given altitude with and without grazing present. The budgets were calculated based on the average over a 10 year period — 1997 to 2006. By using a 10 year period the model can average across inter-annual variability caused by changes in the local weather. The monthly weather at the altitude was calculated based upon the interpolation of climate records from 9 long-term weather stations (Worrall et al., 2009a).

The C budget for a given altitude was first calculated as if there was no management or grazing at that altitude. An ungrazed peat soil at any altitude was assumed to have 1% bare area. This default bare area was in accord with observations of POC fluxes from pristine, vegetated and ungrazed sites (Evans et al., 2006). The impact of grazing upon the proportion of bare soil within the site was based upon the resting and camping habits of sheep. Taylor et al. (1987) report that the daily resting and nightly camping area of sheep represented 1.45% of the area available, therefore, this study increases the area of bare soil by 1.45% per ewe/ha. An increase in the proportion of bare peat soil will have the effect on model predictions by decreasing the primary productivity and would increase POC flux as the model directly correlates POC flux with the proportion of bare peat soil. Increasing proportion of bare soil area was taken as having no effect upon the net ecosystem respiration of CO_2, and fluxes of CH_4, DOC or dissolved CO_2 predicted by the model. The impact of grazing upon the water table was based upon evidence from exclusion studies within the Moor House site, northern England (Worrall et al., 2007a). The study of Worrall et al. (2007a) could only consider the presence/absence of grazers and so the effect of grazers upon the water table does not respond to changes in grazing intensity. The presence of sheep grazing has a significant effect upon the depth of the water table and so the observations of Worrall et al. (2007a) were used in order to derive a monthly adjustment factor for the depth to the water table. The change in the water table predicted by the model will alter the model's predictions of the fluxes of CH_4, DOC and dissolved CO_2.

The outputs from the Durham Carbon Model can be expressed in terms of both a C or a GHG budget. The GHG budget will include the fluvial flux but only that proportion of the POC and DOC exports that was turned over to be released to the atmosphere as CO_2 or CH_4. The dissolved CO_2 was calculated as the dissolved CO_2 present in excess over that which would be present at equilibrium with the atmosphere, and therefore it can be assumed that it would all eventually be released to the atmosphere. The GHG budget would then become

$$F_{CO_2} = PP + R + f_{POC} + f_{DOC} + dissCO_2 + CH_4 + N_2O \quad (1)$$

where: F_{CO2} = the total GHG budget (tonnes CO_2 eq/km^2/yr); PP = primary productivity (tonnes CO_2 eq/km^2/yr); R = net ecosystem respiration (tonnes CO_2 eq/km^2/yr); POC = the annual flux of POC (tonnes CO_2 eq/km^2/yr); DOC = annual DOC flux (tonnes CO_2 eq/km^2/yr); diss.CO_2 = the annual flux of excess dissolved CO_2 (tonnes CO_2 eq/km^2/yr); and CH_4 = the annual methane flux (tonnes CO_2 eq/km^2/yr). f_x = the fraction of the export of component x that is turned over to the atmosphere in streams; all other symbols as defined above but with units as tonnes CO_2 eq/km^2/yr; and N_2O = the annual nitrous oxide flux (tonnes CO_2 eq/km^2/yr). Note that the CH_4 flux was expressed in terms of CO_2 equivalents and as such this study used a greenhouse gas warming potential (GWP) of 23 (Houghton et al., 1996). This study f_{POC} was assumed to be zero in the absence of any other evidence while f_{DOC} = 0.4 (Worrall et al., 2006). Eq. (1) includes N_2O for completeness but this study assumes that grazing makes no effect on such emissions from peat soils.

2.2. Direct losses

The direct losses of sheep grazing were considered as the balance between: consumption of vegetation; production of urine and faeces; respiration of CO_2; fermentation to CH_4; and production of meat and wool. The approach taken by this study was to combine a model of sheep production and consumption (Armstrong et al., 1997a, 1997b) with a model of energy and consumption allocation (Perkins, 1978).

The production of consumable vegetation was based upon the model of Armstrong et al. (1997a, 1997b) and for further model details those studies should be referred to. This uses empirical observations of the growth of four vegetation types from 10 regions across the UK (Fig. 2): *Calluna vulgaris* (hereafter referred to as *Calluna*), *Molinia* sp. (hereafter referred to as *Molinia*), reseeded grasses, and *Agrostis-Festuca* grasses. The biomass relationships predict the growth on a monthly basis and the biomass was adjusted for altitude using a lapse rate with a specific temperature adjustment for the 10 specific regions across the UK. Any hectare of ground was considered as made of any proportion of any of these four vegetation types and may also contain a proportion of vegetation, bare soil or rock that do not provide food for grazers. In this way the available biomass for any hectare at any altitude could be calculated. Live and dead biomass were considered as having differing food value to the sheep. Monthly live production of consumable biomass was considered live for four months and dead for the three subsequent months at which point the consumable biomass moves to litterfall and was removed from the available consumption pool. For the grass species, the proportion of live to dead was directly related to sward height and the sward height was calculated using empirical models derived for each grass type.

The model calculates the amount of biomass required by the sheep. The sheep can be a ewe, a lactating ewe or a lamb; for the purpose of this study all sheep were considered to have been a lactating ewe with a single lamb. It was assumed that lambing will occur in mid April with lambs being removed in mid-August although ewes will

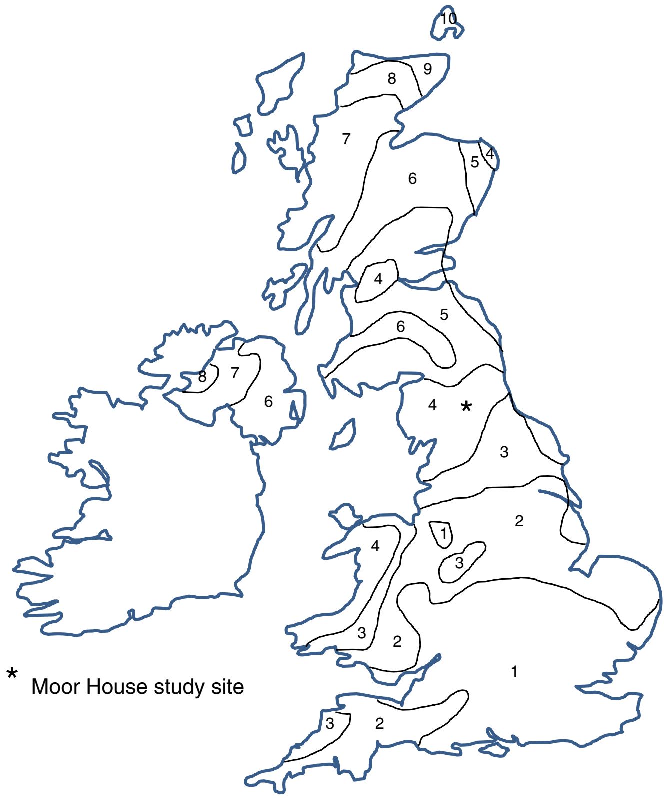

Fig. 2. Location of the climate regions proposed by Armstrong et al. (1997a) based upon the average July temperature. (*) marks the location of the Moor House study site which is the basis for many of the scenarios considered.

stay on the hill and as default they were removed in November after 180 days of being out on the peat ecosystem. It was assumed that daily dry matter intake of a ewe was unaffected by gestation but intake increases during lactation following a quadratic curve that peaks 56 days after lambing at a daily intake 1.44 times that of a non-lactating ewe and returning to normal after 97 days. The lamb was assumed to start eating 6 weeks after birth and the dry matter intake was assumed to increase at 0.8 kg day^{-1} until the point where its weight equals that of a ewe. The lamb live weight was calculated from the live weight of the ewe the previous November and the number of days after birth. Lambs express a preference for consumption of vegetation in the same proportions as ewes.

In the first instance it was assumed that grazing intensity would not exceed the vegetative carrying capacity of the given vegetation scenario at a given altitude, and the carbon and GHG fluxes from grazed peatlands were then calculated. The daily intake from grasses (*Molinia*, *Agrostis-Festuca*, and reseeded grasses) and *Calluna* was considered as the following:

$$I_{DG} = 166.6\,D_G - 43.6 \quad (2)$$

$$I_{DH} = 66.8\,D_H + 2.2 \quad (3)$$

where: I_{DX} = daily dry matter intake per kg of metabolic live weight of sheep for vegetation X, where H = *Calluna* and G = grasses (reseeded, *Agrostis-Festuca* or *Molinia*); and D_X = fractional digestability of vegetation X, where H = *Calluna*; and G = grasses (reseeded, *Agrostis-Festuca* or *Molinia*).

The time taken to consume this intake is:

$$T_{total} = \sum_{i=1}^{n} \left(\frac{1000 j_i}{R_g W_g} \right) \quad (4)$$

where: T_{total} = total grazing time (mins); j_i = the potential daily dry matter intake from vegetation type i if only type i is available (g); R_B = the biting rate (bites/min); and W_B = dry matter mass per bite (mg).

For *Agrostis-Festuca* species the bite rate (R_B) was taken as proportional to the sward height:

$$R_g = 87 - 3.9\ h \quad (5)$$

where: h = sward height of *Agrostis-Festuca* species. For the other vegetation types the bite rate was set as constant. Sward height in grasses was calculated from the predicted standing biomass using empirical linear relationships (Grant et al., 1983).

The bite weight (W_B) of *Calluna* was defined as

$$W_{BH} = 580 M_H 0.86 W_{ewe}^{0.36} \quad (6)$$

where: W_{BH} = bite weight for *Calluna* (mg); and W_{ewe} = live weight of a ewe at the end of season (typically November — kg). The bite weight (W_{BG}) for other grass species, based on Illius and Gordon (1987) was given by:

$$W_{BG} = 860 (R_L M_L + R_D M_D) W_{ewe}^{0.36} \quad (7)$$

where: R_X = the proportion of X in the grass sward where X is L = live matter, and D = dead matter; and M_X = the standing dry matter of type X in the sward where X is L = live matter, and D = dead matter (g).

Burlison et al. (1991) based upon observations from sheep egesta suggested a correction to the above approach that was used in this study:

$$\begin{aligned} &\text{For } (W_{BH} + W_{BG}) > 6.84\ \text{mg} \\ &W_{Bcorr} = 1.125 (W_{BH} + W_{BG}) + 44.7 \end{aligned} \quad (8)$$

$$\begin{aligned} &\text{For } (W_{BH} + W_{BG}) < b6.84\ \text{mg} \\ &W_{Bcorr} = 7.656 (W_{BH} + W_{BG}). \end{aligned} \quad (9)$$

In this formulation of the model, unlike that presented by Armstrong et al. (1997a, 1997b), no correction was made, nor necessary, for overwintering between November and March.Sheep will choose consumable biomass both by its extent (area) and its energy content therefore the actual daily dry matter intake per sheep for each type would be:

$$I_i = I_T \left(\frac{K_i^5 A_i}{\sum_{i=1}^{n} K_i^5 A_i} \right) \quad (10)$$

where: I_i = the actual daily dry matter intake per sheep for vegetation type i (g); I_T = total daily dry matter intake per sheep (g/ewe); K_i = potential daily intake of digestable dry matter of vegetation type i (g); and A_i = proportion of each hectare that is vegetation type i. Note that this equation is to the fifth power and so greatly skews preference toward food value of vegetation without entirely excluding all potential consumable biomass.

The formulation of the model cannot explicitly consider overgrazing but using the above approach the study could calculate a vegetative carrying capacity based upon consumption and loss of biomass up to two limits. First is the limit set by the available dry matter. Second, where the grazing intensity was so great that there simply was not enough time in the day for the sheep to eat the required amount — it is assumed that any sheep can eat for 13 h a day. If either case was met or exceeded by model prediction then overgrazing was assumed and no model prediction could be returned. Equally, the vegetative carrying capacity can be defined as the maximum grazing intensity that does not cause either condition to be met or exceeded.

An alternative approach that this study considered was to estimate a carbon or GHG carrying capacity by which the model was used to predict the grazing intensity at which the total direct and indirect fluxes of C, or GHG, from grazers balance the sink represented by the fluxes of C or GHG that would occur to and from that peat ecosystem under a particular vegetation scenario.

2.3. The fate of consumed biomass

The approach above was used to estimate the amount of vegetation consumed but the fate of this vegetation was also critical for the carbon or GHG balance of a grazed peatland. There are no studies that provide sufficient information on the apportioning of C (or indeed N) species between the various uptake and release pathways, however, several studies have considered whole animal energy and dry matter flows. Perkins (1978) gives the energy balance of a sheep grazing on an *Agrostis-Festuca* grassland as:

$$c = p + r + u + f + m \tag{11}$$

where: c = vegetation consumed; p = production of wool and meat; r = respiration; u = urine; f = faeces; and m = energy lost as methane. The units of this balance could be energy, nutrient or indeed, and as in this case, carbon. Perkins (1978) does not give a carbon balance but does provide an energy balance (Table 1). It was assumed that the carbon consumed from vegetation, any of the available vegetations, could be apportioned in the same manner to the energy for the *Agrostis-Festuca* grassland as observed by Perkins (1978) then it was possible to calculate the fate of the consumed vegetation calculated above by simply using the percentages given in Table 1.

The approach of Perkins (1978) would not cover the emissions of N_2O. The particular risk relative to sheep is the emission of N_2O from manures and in which case this study used emission factors from IPCC (1997). The emission factors are based upon a breeding ewe producing 9.2 kg N/ewe/yr as manure and a lamb producing 3.36 kg-N/lamb/yr. These manure production rates were modified for the proportion of the year that a breeding ewe or lamb was estimated to spend on peat soils. The emission factor for the manure was then 0.02 kg N_2O-N/kg of N in manure and the greenhouse gas warming potential of N_2O was taken as 298 (IPCC, 2007). It should be noted that the lack of detail in IPCC (1997) means that equivalent CO_2 fluxes from N_2O will only vary with grazing intensity but not with region or altitude.

2.4. Peat accumulation foregone

The presence of grazers means that a proportion of vegetation that would contribute to litter and thus would contribute to long term peat accumulation was either converted to forms that are exported from the site (meat, wool, respiration and fermentation) or to forms that are assumed to be more-readily degradable forms of carbon (faeces or urine). This means that peat accumulation would be expected to decline where sheep grazing was present because organic matter inputs to the peat have been restricted.

The peat accumulation was modelled as the first-order exponential decay of both the litter and below-ground dead production. This approach is similar to models of peat depth accumulation as proposed by Clymo (1992). The litter and the below-ground dead production were considered as one and not separated into separate decomposition pathways. The peat accumulation was then the sum of each annual increment which has been allowed to decay over time since it was laid down for each of the species types being considered:

$$C_n^{soil} = \sum_{s=1}^{3} A_s C_s \sum_{t=1}^{n} N_{st}^{lb} e^{-\lambda_s t} \tag{12}$$

where: C_n^{soil} = total peat carbon accumulation (gC/m^2) over n years; N_{st}^{lb} = the total annual carbon input (litter + below-ground dead production) (gC/m^2) for vegetation class s over time t; λ for vegetation class s over time t; A_s = the proportion of the vegetation that is vegetation class s; C_s = the proportional carbon content of vegetation type s; s = vegetation types (for the peat accumulation model 3 vegetation types were considered — *Calluna*, *Eriophorum* or *Sphagnum*) and n = time increment from 1 to 50 years.

The N_{st}^{lb} values can be taken from the biomass prediction described above. The model assumes that the decay constant λ is equal for all peat forming components within a given vegetation type, i.e. there are not separate decay constants for litter and belowground dead production. For the preferred values λ *Calluna* stems and leaves; and *Eriophorum* leaves, have been carried out at the chosen Moor House (550 m asl) over a 23 year period by Latter et al. (1998). Data from Latter et al. (1998) has been re-calculated to give a single value of the first order decay constant for *Calluna* (23 yr^{-1}) and a single value for both *Eriophorum* and *Sphag*num. However, there was no information available that allowed the study to vary λ for different regions or altitudes. The proportion of each vegetation type (A) was taken as that predicted for each scenario considered to be equal for all 3 vegetation classes and so is set at 0.33 — a proportion of vegetation equal to that of the model scenarios discussed below. The carbon content (C_s) was taken as 0.45 gC/(g of dry matter) and was kept the same for all vegetation types.

As when considering the physical impact of sheep above, the peat accumulation foregone in the presence of grazers was calculated as the difference between the annual accumulation of C under grazing and that without grazing for the same vegetation proportions. By examining the difference between the grazed and ungrazed state at any altitude and for each region this study partly mitigates against the problem that there was not enough information available to parameterise the values of λ for different altitudes within the model, however, the biomass values for Eq. (12) do differ with altitude and region.

2.5. Biomass foregone

The flux due to biomass only exists in transition between states of grazing intensity. For example, with grazers present the biomass present on a site will be less than that which would be present if there was no grazing, and therefore, if grazing was completely removed the biomass would increase and this would represent an additional flux of carbon. However, this additional flux of carbon would only occur until a new steady-state biomass was achieved. Therefore, the annual flux due to biomass foregone will be the difference between the biomass present at both states divided by the number of years taken to transition between the two states. The transition between states will be specific to the growth rate of the species present and in order to estimate this the study used the growth curves for upland peat species measured by Forrest (1971), i.e. 20 years to maturity.

2.6. Model scenarios

So as to understand the impact of sheep on the carbon balance of the peat environment we consider one climate zone as defined by

Table 1
The energy balance of a sheep grazing on *Agrostis-Festuca* grassland from Perkins (1978).

Pathway	Energy (kJ/m^2/yr)	Proportion of consumption
Consumption (c)	4659	100
Production (p)	203	4.4
Respiration (r)	1794	38.5
Urine (u)	1940	8.5
Faeces (f)	396	41.6
Fermentation (m)	326	7.0

Armstrong et al. (1997a), i.e. zone 4: the North Pennines (Fig. 2). For this zone we considered grazing intensities from 0.1 to 2 ewes/ha or to the grazing intensity that causes overgrazing as defined above, whichever was the smaller. The model was then run for altitudes between 300 and 900 m — the altitude range within zone 4. Each type of vegetation included within the model was then considered across this altitude and grazing intensity range. The vegetation scenarios considered were based on: *Calluna, Agrostis-Festuca, Molinia*, and reseeded grasses. In addition, this study also considered a vegetation mix typical of the chosen region, the Moor house study site, North Pennines, and northern England (UK national grid reference – NY755330 – Fig. 2) as observed by Forrest (1971) – 33% *Calluna*, 33% *Molinia*; and 33% *Sphagnum* spp. mosses – this mixed vegetation scenario is henceforward referred to as the Moor house scenario. Then in order to assess the biomass foregone the transition between the maximum grazing capacity at an altitude and no grazing at the same altitude was calculated.

2.7. Uncertainty analysis

The model of Armstrong et al. (1997a and b) was taken as reported by the authors, no uncertainty was given to any of the parameter inputs and so no uncertainty assessment could be included within this study. However, in order to assess the variation of predictions across the UK the model was run using the Moor House vegetation combination for the most extreme geographical regions defined by Armstrong et al. (1997a and b), i.e. north-east Scotland and south east England. For the estimation of the uncertainty in the prediction of the physical impact of sheep grazing the Durham Carbon Model was run using the approach to uncertainty estimation of Worrall et al. (2009b) who based on Monte Carlo simulation have reported errors of ±6% for predictions of GHG budgets. Table 2 details the ranges of values used within this study in order to estimate the uncertainty in predictions of the peat mass and biomass foregone. Each model was run 500 times randomly selecting parameter inputs from the ranges defined assuming a uniform distribution within the given range.

All fluxes of all components were judged relative to the atmosphere, e.g. P_g flux is negative. Therefore, a net sink of GHG from the atmosphere would have a negative value. Note that fluvial fluxes can also be judged in the same way and are given a positive value as they are released from the peat soil into the environment. Because grazing intensities are normally presented in terms of ewes/ha then GHG fluxes are explicitly presented in terms of grazing intensity, for example, kg CO_2 eq/ha/yr/(ewe/ha) where the change in the GHG flux (kg CO_2 eq/ha/yr) are expressed in terms of grazing intensity (ewe/ha).

3. Results

The uncertainty analysis for the physical impact and peat mass foregone gave a coefficient of variation of ±15%. The variation between the extreme regions of the UK was between 0.3 and 28% with an average percentage error being 7%. Therefore when considering each vegetation scenario for the North Pennines the average percentage error was taken as ±15% but there is an additional error when considering the results for other regions.

The predicted vegetative carrying capacity of each of the scenarios is shown in Fig. 3 and shows that the carrying capacity of *Calluna* vegetation scenario appears to decline linearly with altitude at an average rate of 0.26 ewes/ha/100 m altitude gain. The other vegetation scenarios show step changes in carrying capacity with the largest carrying capacity being for the *Molinia* vegetation scenario.

The total GHG flux (the physical impact, direct impact and peat mass foregone, but not including biomass foregone because it would only exist in transition) at the maximum predicted carrying capacity at 350 m asl all vegetation scenarios have an approximately

Table 2
The preferred value of inputs and the range used within this study. Values and ranges are taken from Forrest (1971) and Latter et al. (1998).

Input parameter		Preferred value	Preferred range
Decay rates	*Calluna*	0.039	±0.0195
(/yr)	Grasses/sedge	0.043	±0.0215
	Sphagnum	0.043	±0.0215
Litter production at steady-state	*Calluna*	291	±58
(g C/m^2/yr)	Grasses/sedge *Sphagnum*	221	±86
Belowground dead production	*Calluna*	183	±45
(g C/m^2/yr)	Grasses/sedge	48	±22
	Sphagnum	45	±22
Total aboveground biomass	*Calluna*	1778	125
(g C/m^2/yr)	Eriophorum	482	66
	Sphagnum	100	20
Standing dead biomass production	*Calluna*	9	4.5
	Grasses/Sedge	9	4.5
	Sphagnum	9	4.5
Carbon contents	Vegetation	45	5
	Litter	45	5

estimated release of between 171 and 196 tonnes CO_2 eq/km^2/yr (1710–1956 kg CO_2eq/ha/yr — Fig. 4). As altitude increases there are 3 distinct patterns of change. Firstly, the *Calluna* vegetation scenario shows a steady linear decline in line with its carrying capacity while for *Molinia* vegetation scenario there was no decline in emissions and all other scenarios fall in between these two trends. However, because there are differing numbers of ewes at the predicted carrying capacity the decline in emissions at the carrying capacity can be viewed in terms of a GHG efficiency, i.e. the GHG export at carrying capacity divided by the carrying capacity (Fig. 5). In this case most of the vegetation scenarios track each other until an altitude of approximately 650 m asl after which the *Molinia* vegetation scenario was the most efficient and the least efficient was the *Calluna* vegetation scenario. The GHG efficiency is important because it suggests that if the flock size were kept constant at a size that was less than the predicted carrying capacity, then a consideration of efficiency would allow the GHG emissions to be minimised without decreasing the flock size. The differences in GHG efficiency means that if the distribution of vegetation types across an upland peat-covered landscape were known and the flock size also known then the flock could be distributed to minimise GHG emissions without loss of production.

Considering the change in flux with increasing grazing intensity, Fig. 6 shows the variation for each vegetation scenario at 550 m asl (i.e. the height of the Moor House site above sea level). At this altitude it can be seen that increases in grazing intensity have an approximate linear response — an average rate of 1206 kg CO_2 eq/ha/yr/(ewe/ha). As noted in the methodology the estimation of the N_2O emissions does vary with grazing intensity and the variation was from 3.4 kg CO_2 eq/ha/yr at 0.1 ewes/ha to 67.7 kg CO_2 eq/ha/yr at 2 ewes/ha. It can be seen from Fig. 4 that the model predicts a linear response to changing grazing intensity effectively independent of the vegetation type. This result is not too surprising given the assumption of the model, i.e. the model balances the direct fluxes from sheep using the energy balance requirements of sheep and that is not vegetation dependent. Furthermore, the indirect fluxes from sheep are also not vegetation dependent.

It was possible to divide the GHG budget between the fluxes from the ecosystem (soil and vegetation) and the fluxes directly from the sheep itself (respiration etc.). When divided between these two pathways 89% of the total GHG fluxes were lost via the sheep itself while 9% were lost via fluxes from the peat or peat mass foregone. For two scenarios, however, the proportion due to fluxes from peat increased with altitude — that for *Calluna* and Moor House vegetation scenarios.

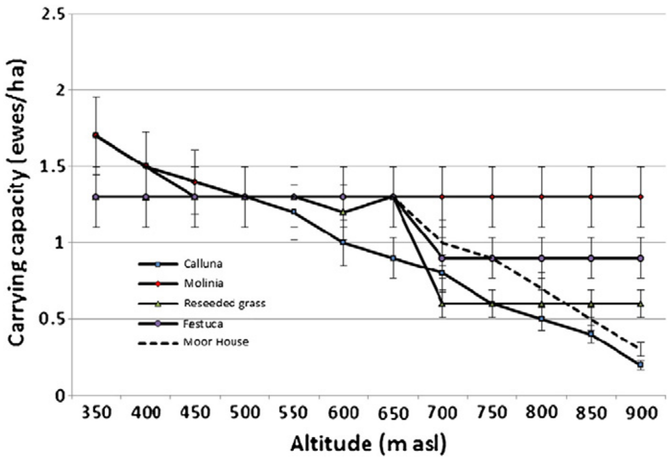

Fig. 3. The vegetative carrying capacity (ewes/ha) for each vegetation scenario considered in this study. The error bar, as described in the text, is ±15%.

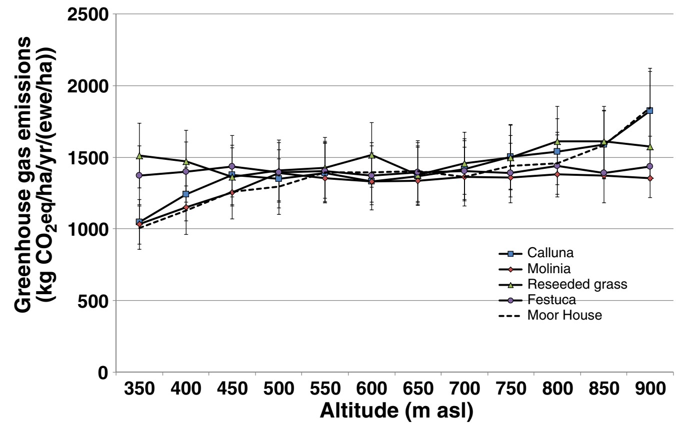

Fig. 5. The greenhouse gas (GHG) efficiency of each vegetation scenario at the predicted carrying capacity. The error bar, as described in the text, is ±15%.

For the other vegetation scenarios there was no significant change with altitude in the proportion of direct to indirect fluxes.

The biomass foregone was estimated as the annual additional sink size that would occur due to regrowth of biomass if the grazing intensity was decreased from the carrying capacity to zero. The biomass foregone was steady for each scenario up to approximately 450 m asl but then declines for most scenarios closely following the carrying capacity for that scenario at each altitude. As for the GHG emissions, the biomass foregone can be expressed as an efficiency per ewe and in which case it can be see that a common value of biomass foregone would be between 53 and 78 kg CO_2 eq/ha/yr/(ewe/ha) with an average value of 71 kg CO_2 eq/ha/yr/(ewe/ha), i.e. this is a small flux compared to the other fluxes calculated above.

The alternative approach to assessing the peat environment's carrying capacity for grazing was not to calculate a vegetative carrying capacity based upon available biomass, but rather to compare the net GHG budget of the environment with and without grazers (Fig. 7). When considering the net equivalent CO_2 budget at the vegetative carrying capacity of the vegetation scenario showed that there were three types of behaviour. Firstly, all vegetation scenarios showed increased losses of net GHG at carrying capacity up to an altitude of 650 m at which point the *Molinia* vegetation scenario showed increasing net losses of GHG with altitude at the vegetative carrying capacity. For the *Agrostis-Festuca* and reseeded grass vegetation scenarios they show a sharp decline in net GHG emissions at altitudes greater than 650 m asl but then rise subsequently (Fig. 7). This sharp change is due to the change in productivity predicted by the underlying model of Armstrong et al. (1997a). The *Calluna* and the Moor House vegetation scenarios both show declines in net emissions at altitudes above 650 m asl and in both cases a sink of net emissions did occur. The variation in prediction of the net GHG budget for both the cases with and without grazers means that when these values come close to each other the variation in the difference between these two budgets becomes amplified; this was observed for the *Calluna* and Moor House scenarios. The carrying capacity can then be recalculated so that the net GHG emissions are not greater than zero. The biggest difference between the scenarios for the GHG fluxes at the vegetation carrying capacity was predicted at altitudes below 650 m asl. The exception was for the *Molinia* scenario where the carrying capacity as defined by GHG emissions continues to decline with increasing altitude (Fig. 8). Where the vegetative carrying capacity varied from 0.2 to 1.7 ewes/ha the GHG carrying capacity varied from 0.2 to 1.2 ewes/ha and the GHG carrying capacity was between 46 and 100% of the vegetative carrying capacity with the largest difference between the two being for the *Molinia* scenario.

4. Discussion

The modelling approach developed here has a number of limitations. Firstly, not all the pathways considered as part of the fluxes from the peat itself can presently be calculated relative to grazing intensity, in particular, the physical impacts of sheep grazing. At present the fluxes of POC and primary productivity respond to changes in grazing intensity, but fluxes of DOC; dissolved CO_2; R_{eco} and CH_4 do

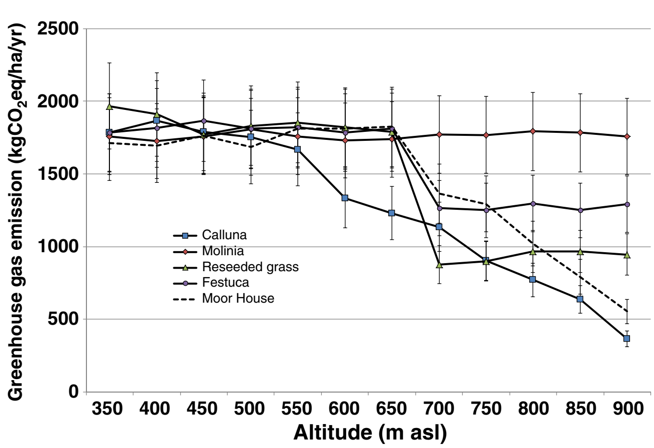

Fig. 4. The greenhouse gas (GHG) budget of each vegetation scenario at the predicted carrying capacity. The error bar, as described in the text, is ±15%.

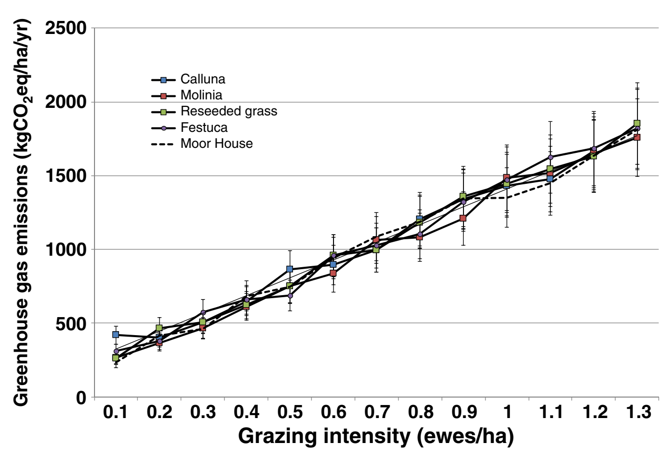

Fig. 6. The GHG budget of each vegetation scenario at 550 m asl for varying grazing intensity. The error bar, as described in the text, is ±15%.

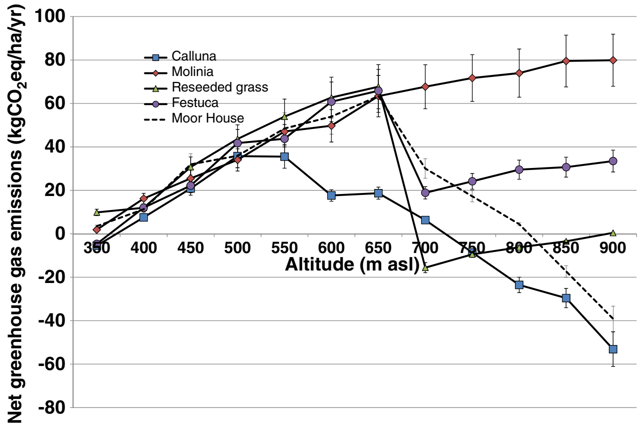

Fig. 7. Net GHG emissions of the ecosystem at estimated carrying capacity for each of the estimated vegetation scenarios considering the comparison of that ecosystem with and without sheep. The error bar, as described in the text, is ±15%.

not currently change with changes in the grazing intensity but rather change only with the presence of grazing. The action of trampling is still not well considered by the model. This study could consider the presence of camping and resting sites relative to grazing intensity but not the impact of trampling outside that ground other than as a presence/absence effect. However, what limits the understanding of the impact of trampling was a lack of knowledge of the trampling behaviour of sheep. The study of Taylor et al. (1987) allows a consideration of the change in the area of resting and camping but not of what area was trampled at what frequency and over what timescale. In addition to the limited data on trampling behaviour, there are even fewer studies on the response of carbon fluxes and stocks to variable trampling intensities. Clay and Worrall (in review) studied the response of gaseous CO_2 to varying simulated trampling intensities and found that although trampling was a significant factor in CO_2 fluxes there was no clear relationship between the intensity of trampling and CO_2 emissions: rather it was the presence or absence of trampling that was important.

For a peat ecosystem that exists because of the accumulation of carbon overgrazing would be true when the peat soil is losing more carbon that it can gain. The study has been able to consider both a vegetative and a GHG carrying capacity. Considering only the fluxes from the peat ecosystem and the peat accumulation foregone (i.e. not the direct losses from the sheep or any biomass foregone due to changes in grazing intensity) over the range of scenarios and at the vegetative carrying capacity shows that the C flux from the peat soil would be between 44 and 240 kg C/ha/yr. Billett et al. (2010) was able to consider 5 studies of 3 sites of which 2 studies were updates of earlier studies. For a pristine Scottish peatland, Dinsmore et al. (2010) found values of −83 and −700 kg C/ha/yr, while for the Moor House catchment, Worrall et al. (2009a) found a range between −200 and −900 kg C/ha/yr. On Bleaklow, in the English Peak District, Billett et al. (2010) give values of between −700 and −1020 kg CO_2 eq/ha/yr. These values for UK pristine peat bogs more than span the range of those values predicted above for grazed peat bogs, and therefore this study predicts that overgrazing of peat soils could occur even at the GHG carrying the capacity of an ecosystems vegetation.

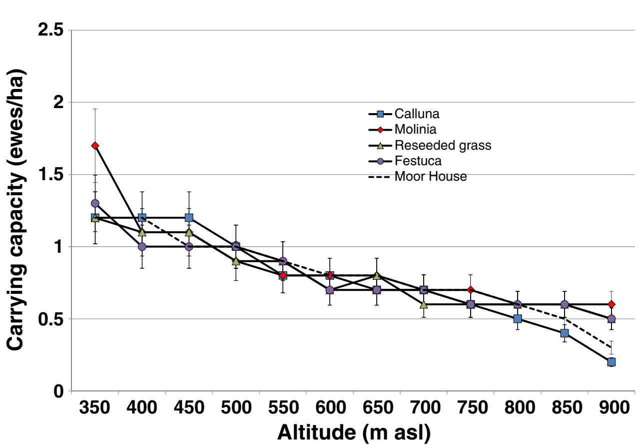

Fig. 8. The GHG carrying capacity (ewes/ha) for each vegetation scenario considered in this study. The error bar, as described in the text, is ±15%.

The implications of the model for grazing management are clear: the model predicts which settings are the most GHG efficient and in which case it is *Molinia* that is the most efficient vegetation type with respect to GHG emissions. Furthermore, it may be possible from these results to assess the GHG impact of conservation interventions. Within the North Pennines region the latest guidelines being used by Natural England advisors come from Natural England (2010a). Recommended maximum annual average stocking rates are suggested as 0.44 ewes/ha to maintain favourable condition, or 0.23 ewes/ha to restore to favourable condition. For a farm in the North Pennines, the stocking densities have declined, prior to 2004 — 0.94 ewes/ha, between 2004 and 2009 — 0.63 ewes/ha; and 2009 to date — 0.56 ewes/ha under a Higher Level Stewardship (HLS — Natural England, 2010b) agreement. Clearly the impact of these changes and these interventions will vary with vegetation and altitude, but for an outline comparison see Table 3. For some of the higher values of grazing intensity these are predicted to be greater than the carrying capacity of the vegetation at that altitude. It is not known exactly over what area or over what flock size these measures have been implemented and so the total impact of these changes cannot be calculated.

The UK government presently only considers a CH_4 emission factor for sheep (8.19 kg CH_4/ewe/yr, 3.28 kg CH_4/lamb/yr – MacCarthy et al., 2010). Given the assumptions of this study, 1 ewe has 1 lamb and that they are on the hill for 180 and 90 days respectively, gives equivalent emissions factors of 13.3 kg CO_2 eq/ewe/yr. However, it is not possible to directly compare this emission factor to total values of GHG export calculated above as these are based upon the export from the peat environment and not the direct emissions from the sheep. However, given the proportions of GHG fluxes direct from the sheep and the proportion of consumption given in Table 1 then it is possible to convert values reported in Fig. 3 into an emission factor for sheep on peat soils and give an average of 598 ± 35 kg CO_2/ewe/yr, as with Fig. 3 this emission increases with increasing altitude for both the *Calluna* and the Moor House vegetation scenarios. If the emission factor is expressed only in terms of CH_4 then the emission factor would be 87 ± 5 kg CO_2eq/ewe/yr. This value is far larger than the emission factor that is currently used. There are a number of reasons for this difference. Firstly, the emission factor calculated here includes the impact on the peat soils as well as the direct emissions from the sheep. Second, it is easy to justify a larger emission factor given the following estimation.

Table 3
The benefit of observed or proposed changes in grazing intensities based upon Natural England (2010a, 2010b) — all values are in kg CO_2eq/ha/yr. Values in brackets refer to altitude range where the change could only refer to a portion of the altitude range included in the study.

Vegetation scenario	Change from maintaining to restoring favourable condition (0.44 to 0.23 ewes/ha)	Change from 2004 to HLS (0.94–0.56 ewes/ha)
Calluna	237 (>800)	315 (>650)
Molinia	214	345
Reseeded grass	277	351 (>650)
Agrostis-Festuca	273	361
Moor House	237 (>850)	364 (>700)

A lamb puts on between 60 and 80 kg of weight in 1 year, that would be between 30 and 40 kg of carbon. Perkins (1978) points out that upland sheep are highly inefficient in terms of their energy balance at about only 5% efficient, i.e. to put on 30 kg of C a sheep must consume 600 kg of C per year, but waste 570 kg of it — given as CO_2 equivalents that is 2091 kg CO_{2eq}. This value would be pro rata lower if grazing intensity were less than 1 ewe/ha. Obviously some of the carbon is returned as dung and urine but equally some is converted to the more powerful greenhouse gas, CH_4. This simple estimation suggests that UK government emission factors are surprisingly low.

5. Conclusions

The study has shown that

i) Vegetative carrying capacity varied from 1.7 to 0.2 ewes/ha over an altitude range of 350 to 900 m asl though the nature of the decline in carrying capacity was dependent upon the exact vegetation type. The greenhouse gas carrying capacity was up to 46% lower than the vegetative carrying capacity.
ii) At the minimum altitude considered GHG fluxes varied between a net source of 171 and 196 tonnes CO_2 eq/km^2/yr at the carrying capacity, this flux declines with altitude but differently for each vegetation leading to differences in GHG efficiency — *Molinia* was found to be the most efficient, i.e. the lowest GHG fluxes for the highest grazing intensity.
iii) For GHG, 89% of the fluxes were directly from the sheep and not from the soil and are therefore not unique to a peat soil environment.
iv) Increases in grazing intensity has an apparent linear response on GHG flux at a rate of 1206 kg CO_2 eq/ha/yr/ewe/ha.
v) The largest flux due to biomass foregone was only 71 kg CO_2 - eq/ha/yr/ewe/ha for each year of a 20 year transition.

The study suggests that emission factors for upland sheep have been greatly underestimated and that in some cases the presently accepted grazing intensities would lead to peatland environments that are net sources of GHG.

Acknowledgements

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References

Armstrong HM, Gordon IJ, Grant SA, Hutchings NJ, Milne JA, Sibbald AR. A model of the grazing of hill vegetation by sheep in the UK .1. The prediction of vegetation biomass. J Appl Ecol 1997a;34(1):166–85.
Armstrong HM, Gordon IJ, Hutchings NJ, Illius AW, Milne JA, Sibbald AR. A model of the grazing of hill vegetation by sheep in the UK .2. The prediction of offtake by sheep. J Appl Ecol 1997b;34(1):186–207.
Ball ME. Floristic changes on grasslands and heaths on the Isle of Rhum after a reduction or exclusion of grazing. J Environ Manage 1974;2:299–318.
Beare MH, Gregorich EG, St-Georges P. Compaction effects on CO_2 and N_2O production during drying and rewetting of soil. Soil Biol Biochem 2009;41(3):611–21.
Billett MF, Charman DJ, Clark JM, Evans CD, Evans MG, Ostle NJ, et al. Carbon balance of UK peatlands: current state of knowledge and future research challenges. Clim Res 2010;45:13–29. (Special Issue).
Burlison AJ, Hodgson J, Illius AW. Sward canopy structure and the bite dimensions and bite weight of grazing sheep. Grass Forage Sci 1991;46(1):29–38.
Burt TP, Gardiner AP. Runoff and sediment production in a small peat covered catchment: some preliminary results. In: Burt TP, Walling DE, editors. Norwich: Catchment Experiments in Fluvial Geomorphology Geo Books; 1984.
Clay GD, Worrall F. The response of CO2 fluxes from a peat soil to variation in simulated sheep trampling. Geoderma in review.
Clay GD, Worrall F, Fraser EDG. Effects of managed burning upon dissolved organic carbon (DOC) in soil water and runoff water following a managed burn of a UK blanket bog. Journal of Hydrology 2009;367(1-2):41–51.
Clay GD, Worrall F, Rose R. Carbon budgets of an upland blanket bog managed by prescribed fire. J Geophys Res Biogeosci 2010;115:G04037.
Clymo RS. Models of peat growth. Suo 1992;43(5–4):127–36.
Dawson JJC, Billett MF, Neal C, Hill S. A comparison of particulate, dissolved and gaseous carbon in two contrasting upland streams in the UK. Journal of Hydrology 2002;257(1–4):226–46.
Dinsmore KJ, Billett MF, Skiba UM, Rees RM, Drewer J, Helfter C. Role of the aquatic pathway in the carbon and greenhouse gas budgets of a peatland catchment. Glob Chang Biol 2010;16(10):2750–62.
Evans MG, Warburton J, Yang J. Eroding blanket peat catchments: global and local implications of upland organic sediment budgets. Geomorphology 2006;79(1–2):45–57.
Evans R. Water erosion in British Farmers Fields — some causes, impacts predictions. Prog Phys Geogr 1990;14(2):199–219.
Evans R. Soil erosion and its impacts in England and Wales. London: Friends of the Earth Trust; 1996.
Forrest GI. Structure and production of North Pennine blanket bog vegetation. J Ecol 1971;59(2):453–79.
Gorham E. Northern peatlands — role in the carbon-cycle and probable responses to climatic warming. Ecol Appl 1991;1(2):182–95.
Grant SA, Barthram GT, Torvell L, King J, Smith HK. Sward management, lamina turnover and tiller population density in continuously stocked Lolium perenne-dominated swards. Grass Forage Sci 1983;38(4):333–44.
Holman IP, Hollis JM, Bramley ME, Thompson TRE. The contribution of soil structural degradation to catchment flooding: a preliminary investigation of the 2000 floods in England and Wales. Hydrol Earth Syst Sci 2003;7(5):754–65.
Hope D, Palmer SM, Billett MF, Dawson JJC. Carbon dioxide and methane evasion from a temperate peatland stream. Limnol Oceanogr 2001;46(4):847–57.
Hope D, Picozzi N, Catt DC, Moss R. Effects of reducing sheep grazing in the Scottish Highlands. J Range Manage 1996;49(4):301–10.
Houghton JT, Meira-Filho LG, Callender BA, Harris N, Kattenberg A, Maskell K. Climate change 1995: the science of climate change. Cambridge: University Press; 1996.
Hynst J, Simek M, Brucek P, Petersen SO. High fluxes but different patterns of nitrous oxide and carbon dioxide emissions from soil in a cattle overwintering area. Agric Ecosyst Environ 2007;120(2–4):269–79.
Illius AW, Gordon IJ. The allometry of food intake of grazing ruminants. J Anim Ecol 1987;56:989–99.
IPCC. (Intergovernmental Panel on Climate Change/Organization for Economic Cooperation and Development). Guidelines for National Greenhouse Gas Inventories. Paris: OECD/OCDE; 1997.
Langland JP, Bennett IL. Stocking intensity and pastoral production. 1. Changes in soil and vegetation of a sown pasture grazed by sheep at different stocking rates. J Agric Sci 1973;81:193–204.
Latter PM, Howson G, Howard DM, Scott WA. Long-term study of litter decomposition on a Pennine peat bog: which regression? Oecologia 1998;113(1):94-103.
MacCarthy J, Brown K, Webb N, Passant N, Thistlewaite G, Murrell T, et al. UK Greenhouse Gas Inventory, 1990 to 2009: Annual Report for submission under the Framework Convention on Climate Change., DEFRA, UK; 2010.
Mackay AW, Tallis JH. Summit-type blanket mire erosion in the forest of Bowland, Lancashire, UK: predisposing factors and implications for conservation. Biol Conserv 1996;76(1):31–44.
Meyles EW, Williams AG, Ternan JL, Anderson JM, Dowd JF. The influence of grazing on vegetation, soil properties and stream discharge in a small Dartmoor catchment, southwest England, UK. Earth Surf Proc Land 2006;31(5):622–31.
Natural England. Higher level stewardship moorland options (HL9, HL10): setting grazing prescriptions (including stocking calendars). Peterborough, UK: Natural England; 2010a.
Natural England. Higher level stewardship — environmental stewardship handbook. 3rd Edition. Peterborough, UK: Natural England; 2010b.
Pengthamkeerati P, Motavalli PP, Kremer RJ, Anderson SH. Soil carbon dioxide efflux from a claypan soil affected by surface compaction and applications of poultry litter. Agric Ecosyst Environ 2005;109(1–2):75–86.
Perkins DF. The distribution and transfer of energy and nutrients in Agrostis-Festuca grassland ecosystem. In: Heal OW, Perkins DF, editors. Production ecology of British Moors and Montane Grasslands, Ecological Studies no. 27. Berlin: Springer-Verlag; 1978.
Robroek BJM, Smart RP, Holden J. Sensitivity of blanket peat vegetation and hydrochemistry to local disturbances. Sci Total Environ 2010;408(21):5028–34.
Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL, editors. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press; 2007.
Taylor JA, Robinson GG, Hedges DA, Whalley RDB. Camping and faeces distribution by Merino sheep. Appl Anim Behav Sci 1987;17(3–4):273–88.
Ward SE, Bardgett RD, McNamara NP, Adamson JK, Ostle NJ. Long-term consequences of grazing and burning on northern peatland carbon dynamics. Ecosystems 2007;10(7):1069–83.
Willatt ST, Pullar DM. Changes in soil physical-properties under grazed pastures. Aust J Soil Res 1983;22(3):343–8.
Worrall F, Armstrong A, Adamson JK. The effects of burning and sheep-grazing on water table depth and soil water quality in an upland peat. J Hydrol 2007a;339(1–2):1-14.
Worrall F, Burt TP, Adamson JK, Reed M, Warburton J, Armstrong A, et al. Predicting the future carbon budget of an upland peat catchment. Clim Chang 2007b;85(1-2):139–58.
Worrall F, Burt TP, Adamson JK. The rate of and controls upon DOC loss in a peat catchment. J Hydrol 2006;321(1–4):311–25.
Worrall F, Burt TP, Rowson JG, Warburton J, Adamson JK. The multi-annual carbon budget of a peat-covered catchment. Sci Total Environ 2009a;407(13):4084–94.
Worrall F, Evans MG, Bonn A, Reed MS, Chapman D, Holden J. Can carbon offsetting pay for upland ecological restoration? Sci Total Environ 2009b;408(1):26–36.
Worrall F, Rowson JG, Evans MG, Bonn A. Carbon fluxes from eroding peatlands — the carbon benefit of revegetation. Earth Surf Proc Land 2011;36(11):1487–98.