Restoration of heather-dominated blanket bog vegetation on grouse moors for biodiversity, carbon storage, greenhouse gas emissions and water regulation:

comparing burning to alternative mowing and uncut management

Final Report

Defra Project BD5104

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Summary and key findings

Over 25% of the UK land area is covered by uplands, the bulk of which comprise blanket bog, dwarf-shrub heath and acid grassland. Blanket bogs are wetlands which are formed under high precipitation and predominantly cool conditions. The high water tables that result, together with acid conditions, suppress decomposition of organic matter whilst also promoting the growth of *Sphagnum* mosses. *Sphagnum* moss is a crucial (but not the only) component for active peat formation, as it increases the water holding capacity of the peat and produces chemicals which suppress decomposition. Long-term blanket bog development in the UK uplands, mainly since the end of the last glacial period, has resulted in extensive peat cover on all but the more steeply sloping ground. This peat accumulation represents a major UK carbon (C) stock which is linked to a range of key ecosystem services, particularly flood prevention, drinking water provision and biodiversity. Although active blanket bogs are a long-term C sink, due to the predominantly anoxic conditions, they have the potential to emit large amounts of methane, potentially causing a net positive contribution to greenhouse gas (GHG) emissions and overall global warming. Importantly, the UK has about 15% of the globally rare blanket bog habitat, containing many specialist species of birds, invertebrates and plants. These upland habitats also attract many visitors and support local economies including through livestock farming and game management.

In England, only around 12% of designated (i.e. Sites of Special Scientific Interest; SSSI) blanket bogs by area are classed as in a favourable condition (according to Common Standards Monitoring Guidance). About 5-15% (between 0.66 and 1.7 million ha) of the UK upland area, and 30% of UK blanket bog, is managed for red grouse by encouraging ling heather (*Calluna vulgaris*) cover. Since around 1850, with the onset of driven shoots, grouse moors have been managed by drainage and rotational burning to encourage heather dominance over other bog vegetation. Whilst grouse moors support local economies, their management has been linked to negative impacts on carbon, water and biodiversity. Moreover, other forms of management (including peat cutting and agricultural use) over several millennia are likely to have reduced the current extent of blanket bog. However, there is relatively little, and sometimes conflicting, evidence on the impacts of management, and particularly grouse moor management, on biodiversity and C storage. Moreover, climate change poses another challenge, as predicted changes in rainfall patterns together with rising temperatures, particularly the increasing frequency of summer droughts, are a potential threat to future bog development. Recently, there have been considerable efforts to reverse blanket bog degradation and increase resilience to climate change impacts through a range of restoration measures including restoring hydrology through blocking and re-profiling drainage 'grips' and gullies, revegetating bare peat, re-introducing *Sphagnum* and other scarce or absent mire species, removing trees and scrub, and increasingly so using alternative mowing management, to encourage 'active' blanket bog vegetation.

However, despite the ecological and economic importance of blanket bogs there is very little evidence on how heather burning or alternative mowing managements alter key ecosystem services of blanket bogs, such as water and carbon storage, and if and how mowing and burning differ in their effects on vegetation composition and structure. There are also few robust datasets available on UK blanket bog C balances, greenhouse gas (GHG) emissions and their controlling factors.

This project aimed to address the key evidence gaps on the effects of alternative management interventions (excluding assessing grip blocking impacts), as identified in a literature review, through a replicated plot-to-catchment scale multi-year study at three sites in northern England, all on heather-dominated blanket bog under grouse moor management. The overall long-term aim of this project was to deliver robust and credible evidence to underpin the development and refinement of possible management techniques as alternatives to burning, for example, applicable through Environmental Stewardship and other agri-environment schemes, to reduce the dominance of *Calluna vulgaris* and support the development of 'active' blanket bog vegetation with a high cover of peat-forming species, particularly *Sphagnum* mosses. However, this long-term aim requires robust evidence based on long-term monitoring well beyond the initial 5-year project phase presented in this report (i.e. covering at least a full management rotation and vegetation re-growth to near maturity). Ideally, long-term monitoring

following initial management would cover the crucial developmental stages of an initial response (1-3 years) and recovery (3-5 years), intermediate transition (5-10 years) and long-term trajectories (10-25+ years). Therefore, the emphasis for the current monitoring phase was to focus on the project's initial objectives of assessing the initial management impacts on biodiversity, carbon storage, GHG emissions and water. For policy relevant input (especially to the government's 25 year plan) clearly the intermediate and more meaningful long-term ecologically aspects would need to be captured by future continuation of research and monitoring.

The three sites chosen for experimental manipulation in this study covered a range of climatic conditions but had a similar average peat depth and plant species composition, and were representative of large areas of *Calluna*-dominated upland blanket bog habitat under grouse moor management, specifically in Britain. The study included a one-year pre-management change period and was designed as a paired catchment study, and compared burning to alternative mowing, with several additional plot-level treatments including brash removal, *Sphagnum* addition and an uncut (do nothing) comparison. Whilst the uncut plots offered a comparison to the managed plots, a 'true' unmanaged control scenario (i.e. catchment) could not be included in this study, but it is acknowledged that such a comparison would ideally be included in assessing potential strategies toward restoring 'active' bog status. Management was carried out initially on areas including the monitoring plots but over subsequent years covered an increasing catchment area as part of a usual management rotation. The experimental results, together with additional laboratory measurements and experiments, and modelled scenarios, allowed impacts on biodiversity (including key upland bird species), hydrology, the magnitude of C budgets and net GHG emissions, and their controlling factors, to be quantified.

The key findings are as follows:

<u>Physical management impacts.</u> Despite an anticipated compaction from the heavy machinery used for mowing, the peat showed resilience and there was no lasting plot-level impact on either peat depth or bulk density at the plots, yet impacts elsewhere, such as from turning and standing machinery, were not assessed. However, mowing did affect the plot micro-topography by removing the tops of hummocks. There were indications that mowing with leaving brash might be beneficial in spreading *Sphagnum* propagules especially at the wettest site. Moreover, burning did not result in the anticipated large peat surface temperature increase compared to uncut or mown plots, identifying a potential methodological issue with previous studies. Ground penetrating radar indicated lower peat accumulation on burnt than on mown plots, but no change in the number of natural peat pipes.

<u>Vegetation assessment</u>. In the immediate post-management period, combined bare, brash and burnt ground cover was higher on burnt than mown plots, but this effect was lost after 4 years. Heather re-growth was slower initially on burnt than mown plots, although after 4 years, heather cover and height was similar on burnt and mown plots. The nutritional value of young heather shoots (more N, P, K, Mn and Mg and less Al) was equally improved by mowing or burning, compared to uncut heather areas, with particular importance to grouse for P and Mn supply. Cotton-grass (*Eriophorum* spp.) cover increased on both burnt and mown plots after management. Specifically, *Eriophorum vaginatum* cover was significantly greater on mown than burnt plots, but to some degree this was also the case in the pre-management period. Cover of total non-*Sphagnum* mosses, particularly *Hypnum jutlandicum* and *Campylopus introflexus*, was greater on burnt than on mown plots, but this difference was partly present pre-management. *Sphagnum* cover was relatively constant across time under all management regimes although there was a sharp increase in total cover on mown plots in the final year which, together with differences in cotton-grass and non-*Sphagnum* moss cover and a more species-specific ecological assessment, indicated a possible different long-term trajectory between burnt and mown plots. Overall plant species diversity was low, decreasing from the wettest to the driest site. The driest site had the highest number of *Sphagnum* species, likely reflecting greater habitat variety, but it also had the lowest overall *Sphagnum* cover with the wettest site by far having the highest cover, as would be expected. The initial relative scarcity of *Sphagnum* at

the drier sites may be an impediment, or delaying factor, in the development of a trajectory towards more 'active' blanket bog with high *Sphagnum* cover, resulting in heather and non-*Sphagnum* mosses regaining dominance (as observed on the driest sites and the burn management after management overall) unless there is a relatively rapid post-treatment 'natural' increase of *Sphagnum,* the re-introduction of *Sphagnum* propagules or additional heather control via repeated mowing. So far, burning appeared to be the least beneficial form of management intervention towards supporting 'active' bog vegetation, particularly at the driest site. The uncut 'do nothing' option showed few, if any, downsides apart from limited recovery of a peat-forming bryophyte layer at the driest site. Mowing regardless of brash management seems to encourage key species and re-establishment of a peat-forming 'typical' bog community, particularly at the wettest site together with an increase in *Sphagnum capillifolium*.

<u>Hydrological impacts</u>. Mown plots had slightly (~2-4 cm) higher (i.e. wetter) water table depths (WTD), as well as higher soil moisture, compared to burnt plots. These effects were particularly apparent in summer and when leaving brash, but there were also considerable site and time period differences. The higher WTD on mown plots were reflected in reduced catchment stream water loss from the mown catchment compared to the burnt catchment at two of the three sites, with mown catchments showing 10% and 20% lower water loss than the burnt catchments in the first and second management cycle, respectively. The site without any observed change in stream water loss also had the least successful burns and had received some previous mowing management in both catchments. Up-scaled flow volumes indicated potentially significant reductions in downstream peak river flooding, estimated to be up to 50 cm under mowing compared to burning scenarios.

<u>Water quality</u>. Mowing resulted in nearly 3 times higher stream phosphorous concentrations than burning during the post-management period. This is likely to be a result of continued leaching from the decomposing brash layer in the mown sub-catchments, and could be important for eutrophication in reservoirs. However, stream nitrogen concentrations, which can also promote eutrophication, were not measured. Both pore water and stream water showed a one unit pH increase over 5 years, which was not significantly affected by management but was partly linked to climatic conditions, in particular increased temperatures. Other stream water quality indicators, including levels of dissolved organic carbon (DOC), varied seasonally and between sites, but did not show any significant impact of management. Peat pore water DOC and UV spectra colour index values (i.e. SUVA) also showed no significant effect of management, but were positively correlated with temperature, and, in the case of SUVA, also with sedge and *Sphagnum* cover, whilst negatively correlating with heather cover. This highlights the need for wider catchment-scale and process-level assessments covering topographic and vegetation complexity.

<u>Cranefly emergence, abundance and bird population modelling</u>. The increased surface peat moisture in mown compared to burnt areas in the dry year after management, resulted in higher cranefly emergence. However, cranefly emergence on mown areas was reduced in the following two wet years. Cranefly emergence was also consistently lower on the wettest site, particularly in the wetter mown plots. These findings are likely to reflect a lower and upper soil moisture limit for optimum cranefly emergence between 80% and 95%. Cranefly abundance on transects was overall higher in mown than burnt catchments. However, the upper 95% limit, cranefly species and potential food source switching by birds were not considered in current predictive models. The modelled implications for golden plover fledging production showed that numbers would be higher in mown than burnt areas, this effect being strongest in the relatively dry year of 2014 when cranefly abundance was lowest. Modelling of the effects of drier summers more likely under climate change, based on the cranefly emergence and soil moisture data, predicted a greater resilience to future drier summers of upland bird numbers (i.e. dunlin, golden plover and red grouse) under mowing, particularly when leaving brash, than under burning. However, the potential that mowing might make generally wetter sites too wet for cranefly larvae survival (i.e. lower emergence) and more detailed changes in plant species composition (specifically key ecological species) in response to environmental changes and micro-topographic management impacts on nesting preferences by birds (i.e. importance of hummocks for dunlin) were not included in the model.

Soil C cycling and decomposition. Soil respiration rates from decomposition processes were not significantly influenced by management intervention in the field. However, such field measurements capture fluxes from the whole peat column, whereas management intervention is most likely to affect only the peat surface layers. Therefore additional measurements and experiments were conducted on surface peat under controlled, laboratory conditions. These showed that decomposition rates in the surface 5 cm were lower on burnt than on mown plots, with brash left. The temperature sensitivity (Q_{10}) of decomposition in the surface 5 cm of peat was also lower on burnt than mown with brash left plots, as was the response to variation in soil moisture. In addition to loss of biomass (via combustion) otherwise available for decomposition and reduced plant-derived labile C inputs available to microbes, charcoal input from burning was identified as a potential additional mechanism explaining this difference. Charcoal was linked to an increase in bulk density, with subsequent possible negative effects on microbial activity and hence lower decomposition. Heather associated mycorrhizal fungi were shown to be able to break down very old peat carbon and hence could limit the longevity of the peatland carbon store, whilst potentially also increasing DOC concentrations and hence reducing the quality of water draining peatlands.

Ecosystem CO_2, fluvial C budgets and methane. Annual net CO_2 flux budgets based on net ecosystem exchange (NEE) chamber CO_2 fluxes for uncut plots showed that 5-year mean C gain (with considerable inter-annual variation) was greatest at the wettest site, but that the driest site was a small net C source. Both burnt and mown plots switched from a net C sink to a net C source after management. Net annual C losses were greater from burnt than mown plots in the year following management. However, 4 years after management intervention, C losses from burnt (excluding losses during combustion) and mown plots, averaged across the three sites, were very similar. Overall C export as DOC was about 13 times higher than as particulate organic carbon (POC), but management did not cause any significant change in either DOC or POC stream export rates. Both DOC and POC export rates showed a high seasonal variability and a positive correlation with temperature and annual C budgets highlighted the need to include POC fluxes for the 'modified' peatland category in the IUCN UK's Peatland Code. Methane (CH_4) fluxes increased with higher water tables, and showed a weak positive effect of sedge cover and soil temperature. Methane fluxes were much higher in the last two years of the study, particularly in 2016 and at the wettest and less modified site, and higher fluxes corresponded to higher than average mean annual soil temperatures and water tables. There was no difference between methane fluxes from non-vegetated and vegetated areas of burnt and mown plots. However, methane emissions were higher from vegetated areas of uncut plots than from burnt plots, but did not differ from mown plots. Overall, plant mediated transfer contribution (i.e. via sedges) to methane emissions (as compared to soil only emissions) was around 60%.

Net ecosystem C balance (NECB) and net greenhouse gas (GHG) emissions. The mean annual NECB estimate for each site (based on proportionally managed area) indicated an overall small C loss across all sites, although values varied greatly between sites and years, primarily due to variation in net ecosystem exchange of CO_2 (NEE) and methane fluxes. However, when median fluxes (a more robust measure of central tendency) were used instead of mean fluxes, annual NECB losses reduced, and there was a small C gain at the wettest site. Given that all three sites are located in north-west England and are therefore subject to similar climatic conditions, and that all three sites are *Calluna* dominated grouse moors, the fact that NECB value can be either positive or negative for different sites in the same year suggests that there are likely to be unmeasured, or even unknown, factors influencing the C dynamics on these peatlands.

After management intervention, median NECB values showed C losses under both mowing and burning scenarios that were on average 8 times larger than the C gains of the uncut scenario, primarily due to lower values of NEE. The size of the net C source from the burning scenario was higher than from the mowing scenario, once the C loss during the burning of the heather biomass itself was taken into account. However, there remains uncertainty in these NECB estimates, because long-term brash decomposition losses for mowing could not yet be considered and fluvial C losses included losses from areas with little peat cover, also including erosion from slopes, whereas CO_2 and methane fluxes were measured on predominantly flat areas of deep peat.

The up-scaled estimates for net GHG emissions (including CO_2, CH_4 and N_2O) expressed as CO_2 equivalents (CO_2-eq), also differed considerably between sites. The average net GHG emissions for the uncut scenario were positive (i.e. a net source) at the driest and wettest sites, but negative (i.e. a net sink) at the third with intermediate mean annual water table depths. Mean net GHG emission values under the burnt and mown scenarios were similar; they were all positive and much greater than the uncut scenario. Despite the high global warming potential (GWP) of N_2O, the inclusion of N_2O emissions had little influence on the net GHG emissions, but inclusion of long-term methane fluxes had a large effect; as for NECB, net GHG emissions varied considerably, depending on whether the mean or the more robust median fluxes were used, especially affecting methane emissions. During the study period, the results of the uncut plots for the two drier sites agreed fairly well with current assumptions of the IUCN UK's Peatland Code for modified and drained blanket bog sites. However, on the wettest and less modified site, the final two years resulted in much higher net GHG emissions (i.e. the Peatland Code seems to be underestimating the net GHG emissions), mainly due to very high methane emissions which could be linked to warmer and wetter conditions. These findings call for caution against the assumption of a "the wetter the better" management approach within a blanket bog GHG emissions context, as overall ranges in methane emissions between sites and over time indicated a possible water table depth threshold in relation to achieving both, a net C sink and a beneficial net GHG balance.

Peat C sequestration. C accumulation rates based on dated peat cores for all three field sites under burn rotation management were similar to a previously reported estimate for unburnt management (as part of a burn comparison) when compared over the same time period. Whilst chamber CO_2 fluxes suggested an overall larger net loss of C on burnt than mown plots when losses from burnt biomass were included, C stock changes in peat cores indicated that recent peat C accumulation was actually higher under more frequent burn rotation. This discrepancy between C sequestration estimates based on the flux chamber method and those based on the peat stock inventory approach highlights the importance of long-term C stock assessment. It also highlights the importance of quantifying processes that are not captured by the flux approach, such as charcoal inputs bypassing decomposition and the inclusion of decomposition fluxes arising from deeper and older peat layers. A previously published study, based on re-sampling of sites, has suggested that peat soils in the UK have lost C from their surface layers over recent decades. However, laboratory and field studies carried out as part of this project demonstrate that short-term peat expansion and contraction, related to seasonal moisture changes, and associated bulk density changes, could possibly explain these apparent C losses.

Past management impacts. Model scenarios for Moor House (the most intensely studied UK blanket bog site) over the past two centuries were linked to palaeo-ecological WTD reconstructions. These scenarios revealed drainage and burn impacts from past grouse moor management on peatland functioning (i.e. lowering the water table and less carbon input), reducing carbon storage and methane emissions. Model scenarios over a period of 8,000 years were constructed to provide an otherwise unavailable comparison of an unmanaged peat development scenario and account for the effects of a range of historical land management, including agricultural cultivation, peat cutting, and biomass (i.e. burning, harvest and grazing) removal. Modelled scenarios for six UK locations indicated large impacts of historical management on soil C stores and peat areas. The combined historic management impacts indicated a vast potential loss of soil C (more than 50% across the test areas), including from blanket bog and other likely former peat areas, during the last 2,000 years. These effects were greater in low-lying areas close to population centres than in remote rural upland areas. Therefore, there is considerable potential from peatland restoration and reverting to land management practices that allow long-term soil C sequestration and peat formation (particularly in areas of high past soil C losses), subject to future climatic conditions being suitable.

Cost and emissions analysis. An analysis of costs and emissions associated with management interventions to date showed that, for the three study sites, costs for mowing (including sub-contractor and gamekeeper costs) were more than six times those for burning. Although mowing required about half the man hours that burning required, about 10 times the amount of fuel was used, resulting in 3 times higher fuel combustion CO_2 emissions per hectare during mowing. However, when direct CO_2 emissions from heather biomass burning were factored in,

overall CO_2 emissions from burning were 70 times greater than those from mowing (the latter not including long-term brash decomposition). A similar result was found for other air pollutants (e.g. particulates and NOx), emissions from heather biomass burning also being much higher than those from mowing-related vehicle emissions.

Synthesis of effects of management interventions to date. A detailed cost-benefit analysis (CBA), including long-term effects on ecosystem services, was not possible at this stage, as the experimental plots are still in a transition period and catchment management is not completed. However, a summary matrix (see Table 29 at the end of this report) was constructed of the initial management impacts to date on all major measured ecological parameters in relation to ecosystem services and the aim of supporting 'active' blanket bog development and reducing heather dominance through either mowing or burning. Overall, based on 30 parameters, mowing was marginally more beneficial than burning. Mowing had positive effects on 8 parameters, compared with 6 for burning, and had negative effects on 7 parameters compared with 11 for burning, had 14 'no change' effects compared to 13 for burning, and one category was not assessed for mowing. In particular: mowing tended to support the development of more 'active' bog vegetation, although heather regeneration was not different to burning; mowing did not cause any significant negative impacts on peat properties; mowing did not cause any overall change in water quality, although there was an indication of possible contribution to eutrophication (i.e. in reservoirs but N was not measured) from decomposing brash; mowing raised the water table and increased soil moisture with positive impacts on bird populations, especially in future drier summer climate scenarios; mowing reduced stream flow rates; and mowing led to lower carbon flux losses, lower net GHG emissions, and less air pollution than burning, when including biomass combustion losses. However, for all these effects, inter-annual and site variability was considerable and long-term trajectories remain unknown. It is noteworthy that the current study focused on comparing different management representative of only about 30% of British blanket bog (mainly the Pennines); ideally a 'no management' scenario would be included at the same plot-to-catchment scale monitoring level in any future assessment.

In summary, the findings from the different elements of the project are to be seen as preliminary (as so far only four years of post-management monitoring have been done), which indicate that mowing could be an appropriate, albeit more costly, alternative to burning of heather dominated blanket bog on grouse moors, benefiting key ecosystem services, particularly related to hydrology, and potentially encouraging development of 'active' blanket bog vegetation. However, process-level effects, and the associated long-term impacts on ecosystem services, of a complete change in catchment management practice and vegetation re-growth require time to develop, particularly in cold, wet and thus slow growing upland ecosystems. Overall, comparisons between the two types of management highlighted the need for continued management and monitoring over at least a complete management cycle (requiring ca. 10-15 years), possibly together with additional plot-level treatments like repeated mowing, as the long-term (and thus robust and policy relevant) impacts are not yet adequately captured or predictable. For example, damage to *Calluna* can be expected more readily when the plant is under stress from, for example, rising water tables as was achieved by mowing; additional years should reveal if this will lead to a general decline in *Calluna* cover as indicated by frost and heather beetle damage at the two wetter sites. Required time periods for a continuation of monitoring towards providing such robust long-term policy relevant evidence on key ecosystem parameters (based on catchment rotational management, interannual climate variability and vegetation growth rates and plant community development) can be estimated to require between 10 to 25+ years depending on the parameter, e.g. for C budgets (10+ years), methane emissions (15+ years), water budgets (20+ years) and vegetation dynamics and biodiversity (25+ years). Finally, ideally a catchment-scale 'no management' scenario should be considered in future research as another available management option.

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List of acronyms used in the report

ATV – all terrain vehicle

AWS – automated weather station

BACI – Before After Control Impact

BD – bulk density

CBA – cost benefit analysis

CI – confidence interval

CUE – carbon use efficiency

Defra – Department for Environment, Food and Rural Affairs

DOC – dissolved organic carbon

EC – eddy covariance

ECN – Environmental Change Network

E4/E6 – UV465/UV665 (water quality measure based on UV absorption spectra)

FTIR – fourier-transform infra-red spectroscopy

GC – gas chromatograph

GHG – greenhouse gas

GIS – geographical information system

GPP – gross primary productivity

GPR – ground penetrating radar

GWP_{100} – global warming potential of a greenhouse gas over a 100 year time-fame relative to that of CO_2 (CO_2-eq)

ICP – inductively coupled plasma

IUCN – International Union for Conservation of Nature

LAI – leaf area index

LCP – light compensation point

LOI – loss on ignition

MAT – mean annual temperature

MAP – mean annual precipitation

NE – Natural England

NEE – net ecosystem exchange (of CO_2)

NEEmax – maximum NEE (under full light)

NEE500 – NEE at a PAR of 500 (µmol m^{-2} s^{-1})

NEBP – net ecosystem biome productivity

NECB – net ecosystem carbon balance

NMR – nuclear magnetic resonance spectroscopy

NNR – National Nature Reserve

NPP – net primary productivity

NSRI – National Soil Resources Institute

NVC – National Vegetation Classification

PAR – photosynthetically active radiation

POC – particulate organic carbon

Py-GC/MS – pyrolysis gas chromatography mass spectrometry

Q_{10} – temperature sensitivity (increase in respiration with a 10°C rise)

R_a – autotrophic (root) respiration

R_{ab} – above ground respiration (shoots)

RDA – redundancy analysis

R_{eco} – ecosystem respiration (all respiration components)

R_h – heterotrophic respiration (from decomposition only)

R_r – root respiration

R_s – total soil respiration (from roots and decomposition)

SCP – spheroidal carbonaceous particles

SE – standard error

SM – soil moisture

SOC – soil organic carbon

SR – soil respiration (flux)

SR_c – heterotrophic respiration (from cut, root free areas)

SQRT – square root (transformation)

SSSI – Site of Special Scientific Interest

STDEV – standard deviation

SUVA – specific ultraviolet absorbance (water quality measure of UV absorption at 254 nm/DOC)

T_{air} or T_{soil} – air or soil temperature

UGGA – ultraportable greenhouse gas analyser

UV – ultraviolet light (absorption wavelength in nm)

WTD – water table depth

List of site, field experimental and management abbreviations

Nidd (N) – Nidderdale

Moss (M) – Mossdale

Whit (W) – Whitendale

C – control (i.e. burnt) catchment

T – treatment (i.e. mown) catchment

Pre(-period) – before April 2013 (i.e. before first management intervention)

Post(-period) – after April 2013 (i.e. after first management intervention)

DN – uncut (do nothing)

FI – burnt (fire)

M – mown (cut)

LB – (mown) left brash

BR – (mown) brash removed

Sp – *Sphagnum*

±brash – with or without brash (removal)

±Sp – with or without *Sphagnum* pellet addition

NEE_{new} – NEE (plots) with regrowing vegetation

List of common chemical symbols and abbreviations used in this report

Al – Aluminium

BC – Black carbon (air pollution - subcomponent of $PM_{2.5}$ and PM_{10} see below)

C – Carbon

C_{org} – Organic carbon (soil)

^{14}C – Radiocarbon

Ca – Calcium

CH_4 – Methane

CO_2 – Carbon Dioxide

C/N – Carbon to Nitrogen ratio

Cu – Copper

Fe – Iron

K – Potassium

Mg – Magnesium

Mn – Manganese

N – Nitrogen

Na – Sodium

NH_3 – Ammonia

NMVOC – Non-methane volatile organic compounds

NO_x – Nitrogen oxides most relevant for air pollution [nitric oxide (NO) and nitrogen dioxide (NO_2)]

N_2O – Nitrous oxide

OC – Organic carbon (air pollution - subcomponent of $PM_{2.5}$ and PM_{10})

P – Phosphorus

Pb – Lead

PM_{10} – Coarse particulate matter (2.5 – 10 µm)

$PM_{2.5}$ – Fine particulate matter (<2.5 µm)

SO_2 – Sulphur dioxide

SO_4 – Sulphate

VOC – Volatile organic compound

Zn – Zinc

1. Background, aims and objectives

1.1 Background

Over 25% of the UK land area is covered by uplands (Haines-Young et al., 2000), the bulk of which comprise blanket bog, dwarf-shrub heath and acid grassland. Blanket bogs are wetlands forming under high precipitation and predominantly cool conditions, where high water tables and acid conditions suppress decomposition and favour *Sphagnum* moss growth and 'active' peat formation (JNCC, 2006) such that long-term bog development results in extensive peat cover on all but the more steeply sloping ground. This peat accumulation of active bogs in the uplands represents a major UK carbon (C) stock which is linked to a range of key ecosystem services (Haines-Young & Potschin, 2008), particularly flood prevention, drinking water provision and biodiversity (Harper et al., 2018). Although active blanket bogs are a long-term C sink, they have the potential to emit large amounts of methane (Baird et al., 2009), potentially causing a net positive contribution to greenhouse gas (GHG) emissions. Importantly, the UK has about 15% of the globally rare blanket bog habitat (Evans et al., 2006), containing many specialist species of birds, invertebrates and plants. These upland habitats also attract many visitors and support local economies including through livestock farming and game management.

About 5-15% of the UK upland area (Grant et al., 2012), and 30% of UK blanket bog (Natural England, 2010), is managed for red grouse by encouraging ling heather (*Calluna vulgaris*) cover. Since around 1850, with the onset of driven shoots, grouse moors have been managed by drainage and rotational heather burning to encourage heather dominance over other bog vegetation and their management has been linked to adverse conditions of blanket bogs (Natural England, 2008) with negative impacts on carbon, water and biodiversity. In England, only around 12% of designated (i.e. Sites of Special Scientific Interest, SSSI) blanket bog by area is now classified as in a favourable condition (Natural England, 2015). However, there is relatively little and sometimes conflicting evidence on the impacts of management (particularly grouse moor management) on water quality, C dynamics (i.e. storage, fluxes, budgets) and biodiversity (see a review by Harper et al., 2018).

Moreover, climate change poses another challenge, as changes in rainfall patterns with an increasing frequency of summer droughts, are a threat to future bog development. Recently, there have been considerable efforts to reverse this degradation and increase resilience to climate change impacts by blocking drainage grips and using alternative mowing management to encourage 'active' blanket bog vegetation. Despite the ecological and economic importance of blanket bogs, few robust data are available on UK blanket bog C balance, net GHG emissions and their controlling factors, and on how heather burning or alternative mowing managements affect C storage and other key ecosystem services, and if and how mowing differs in its effects on vegetation composition and structure compared to burning and leaving bog 'unmanaged'.

This study was commissioned by Defra in 2012 in order to address identified evidence gaps regarding the impacts of current and possible alternatives to burn-rotation management of heather dominated blanket bog on carbon, water and biodiversity.

1.2 Aims and objectives

The overall long-term aim of project **BD5104** '*Restoration of blanket bog vegetation for biodiversity, carbon sequestration and water regulation*' was to deliver robust and credible (long-term) evidence to underpin the development and refinement of possible management techniques, for example, applicable through Environmental Stewardship and other agri-environment schemes, to reduce the dominance of ling heather (henceforth referred to as heather) and support the development of 'active' blanket bog vegetation with a high cover of peat-forming species, particularly *Sphagnum* moss species. This required an initial screening for the most suitable management techniques, which were then to be included as part of a long-term and robust manipulative

field experiment to acquire experimental data within a realistic land management context. This would provide scientifically sound and meaningful data upon which to base policy advice and subsequently inform management decisions, considering both environmental and socio-economic implications. Given the relatively slow rate of change in blanket bog vegetation, and recovery of heather (and other vegetation) from management (see Hancock et al., 2018 and Harper et al., 2018), it was anticipated that the initial phase had to focus on the short-term objectives of assessing the initial re-growth in addition to the assessments of initial carbon, water and biodiversity impacts. However, the study was designed to allow addressing the project's long-term aim, although this would require an extension well beyond the initial 5-year project period.

BD5104 covered an initial 5-year period and was divided into four parts; the main aims from objectives for each part are summarised below.

1. To conduct a ***literature review*** of potential techniques to address heather dominance and help promote 'active' peat-forming vegetation, particularly *Sphagnum* species, on blanket bog, and to use the results of this review, together with a stakeholder workshop at the start of the project, to identify practical management options for experimental field testing.
2. To ***field test*** the identified management options within a fully replicated long-term field experiment, with plot-level and catchment-scale replication, and to monitor the impacts on key ecosystem services related to carbon, water and biodiversity, specifically:

   a) plant species composition, including bryophytes, and likely impacts on peat formation;
   b) water table and stream flow;
   c) carbon fluxes from peat in fluvial and gaseous forms;
   d) peat micro-topography and peat pipes;
   e) soil moisture and cranefly abundance.

3. To ***evaluate the impact of treatments*** on vegetation dynamics, stream flow, water budgets, carbon stocks and fluxes, greenhouse gas emissions, peat properties, cranefly and related bird abundance, and, when appropriate, upscaling findings in space and time based on statistical and process-based models.
4. To ***provide data toward a cost-benefit analysis*** to determine the cost of achieving the management in relation to a range of ecosystem services.

However, whilst there were clear time constraints in delivering evidence in relation to the long-term aim (i.e. reducing heather domination and supporting active blanket bog vegetation), the experimental platform was designed to address current evidence gaps (specifically those highlighted by Harper et al., 2018) by providing crucial and robust short-term background data (from the initial phase) in addition to more valuable long-term evidence (through an extension), which could, of course, also be complemented with additional work at other sites by other projects. Whilst the project's focus was limited to mainly comparing burning versus alternative mowing management, future projects could be particularly valuable in addressing a potential 'no management' option in similar detail.

1.3 Field and laboratory studies

To complement the main experimental work aspects, additional field and laboratory studies were undertaken to investigate specific knowledge gaps identified during the literature review, stakeholder workshop and the associated PhD project. These included studies of: moisture impacts on peat shrinkage/expansion rates; plant volume impacts on chamber carbon flux measurements; peat decomposition rates in relation to temperature and moisture; potential mycorrhizal priming aspects; past burn frequencies; UV water quality measures; and the chemical composition of the peat in relation to decomposability and model representation.

2. Summary of literature review

A detailed literature review on ”potential techniques to address heather dominance and help encourage appropriate ‘active’ *Sphagnum* supporting peatland vegetation on blanket bog and identify practical management options for experimental testing” was conducted throughout 2012. A first draft was produced by November 2012 and a final revised version in August 2017 (provided in **Annex A**). The revisions also included feedback on the proposed management techniques to be tested in the field experiment which was provided at a stakeholder workshop in February 2013 (see **Annex A.1**), before the management treatments commenced. The review (**Annex A**) assessed current knowledge of management options with respect to the reduction of heather dominance to aid the restoration of ‘active’ blanket bog (in addition to the ongoing infilling of drainage channels called grip-blocking right across the UK), particularly regarding the restoration of hydrological function and *Sphagnum* growth; the focus being on those options of most relevance to the management of heather dominated blanket bog vegetation. The review examined grazing, burning and mowing, with a brief consideration of herbicides and other potential management options. The possible effects of management techniques on ecosystem services, such as greenhouse gas emissions, were also considered. The report concluded with recommendations for an experimental design; this excluded herbicides (deemed inappropriate as blanket bog locations are often within SSSIs) and grazing (due to complexities around variability and practicalities regarding time scales). Although a ‘no-management’ option was not considered in general (i.e. due to build-up of combustible biomass and thus increasing fire risk), it was considered as a specific uncut plot-level comparison to both, mowing and burning.

2.1 Grazing

Traditional low-intensity summer grazing has little effect on vegetation composition and might even help to maintain the balance of species present in bog communities (O’Brien et al., 2007). Excess grazing, and associated trampling and uprooting, can lead to the loss of slow growing heather but also peat forming moss species (English Nature, 1996), whilst increasing cotton-grass species (*Eriophorum* spp.) and other graminoids (particularly *Molinia* and *Trichophorum*) and/or leading to an increase in bare ground and erosion (Martin et al., 2013). There is contradictory evidence on the relationship between grazing and enrichment of the substratum. Supplementary feeding may introduce more nutrients into the system, as defecation and urination can raise the soil’s nutrient status, although the effects are likely to be localised (RSPB, 1995 in Shaw at al., 1996). Consequently, this increased nutrient input (likely benefitting fast growing species more than slow growing bog species) needs to be considered as part of any grazing scheme. However, excluding grazing does not necessarily lead to less heather domination, at least not in the short term. In a 15-year sheep grazing exclosure experiment carried out on poor quality blanket bog (species poor, little cover, active erosion) at Moor House, major changes in species composition, pattern of vegetation and structure were observed, with *Calluna* growing, from an early position of scarcity, to dominate parts of each exclosure (Rawes, 1983).

Recommendations on stocking densities are confusing and require more research (O’Brien et al., 2007). Backshall et al. (2001) provide guidelines, including recommended stocking rates, for the sustainable grazing of blanket mire. Generally, the wetter the site the lower the productivity of the blanket mire plants and the greater the sensitivity to grazing (Coulson et al., 1992). It is noteworthy that heather is particularly susceptible to grazing damage in the spring and autumn/winter. High levels of grazing can also reduce the cover of *Sphagnum* and lichens, whilst cotton-grass and/or purple moor-grass (*Molinia caerulea*) may increase in dominance (Coulson et al., 1992; Rawes and Hobbs, 1979). However, light grazing by sheep, without burning, is likely to be an acceptable management for blanket bogs in the interests of conservation (Rawes and Hobbs 1979); light autumn/winter grazing during plant establishment phases and summer grazing during regrowth could be considered as ways of weakening heather over time, thus facilitating recovery of slow growing ‘active’ peat-forming bog species. For

year-round sheep grazing, Hulme and Birnie (1997) calculated that grazing levels of between 0.5 and 1 sheep ha^{-1} are the maximum levels permissible if blanket bog vegetation is not to suffer damage. However, the specific effects of grazing on peat soils are complex and depend on a number of factors as outlined in Ausden and Treweek (1995 cited in Backshall et al., 2001), the literature review and the stakeholder workshop highlighted that it would be difficult to adequately address such complexities in the project's experimental design.

2.2 Burning

Burning has been used to manage upland vegetation in Britain for centuries, principally for stimulating new growth of grasses or heather. However, large scale and repeated heather burn rotations (typically on patches of about 0.5 – 1.0 ha on a 10-15 year rotation) have only been introduced across the UK during the past 200 years (e.g. Hay, 2012) in relation to grouse management (including muirburn in Scotland). Careful periodic burning of upland vegetation can have advantages for agriculture, game rearing, wildlife conservation and intrinsic landscape appeal. Overall, burning effects on vegetation depend on many factors but, in general, increase species diversity and richness on dwarf shrub heath (Grant et al., 2012); however, findings might not be applicable to areas managed as grouse moors and relatively few studies have assessed the effects on blanket bogs (Grant et al., 2012).

Burning management of any kind may be detrimental to blanket bog but the most severe damage is likely to occur during uncontrolled fires where considerable biomass (fuel) has been allowed to build-up (as could be seen in the 2018 UK moorland fires, particularly at Saddleworth near Manchester), which a careful regular burning regime might prevent (Yallop et al., 2006). However, fire intensity is highly variable and wildfires are not necessarily more damaging than controlled burning (Clay et al., 2010a). The initial vegetation composition (and hence the previous management history) can influence the fire intensity and, in old stands of heather, fires may be particularly intense. However, large stands of woody heather (i.e. rank heather) or extensive areas of dense grass litter may develop in the absence of burning and grazing, and these can then pose a considerable fire hazard. In situations where scope for alternative management is limited, controlled and careful burning has been used to reduce the available fuel load, to create fire breaks and to generally reduce the likelihood, severity and extent of uncontrolled fires. Moreover, seed longevity varies between species and long-lived seeds from *Calluna* will probably benefit more from a long burn rotation than the short-lived seeds of many other moorland species (Hobbs et al., 1984).

The intensity and frequency of a fire is critical. For example, damage to bryophytes can often be avoided if a fast-moving 'cool' fire occurs when the ground is frozen (Rowell, 1988). Some species, including some, but not all, *Sphagnum* species, are capable of rapid re-colonisation following burning (Daniels, 1991 cited in Backshall et al., 2001). Generally, low intensity fires on blanket bogs may have little long-term effect on *Sphagnum* species, but moderate to high intensity fires can eradicate them. However, results in relation to burn rotation length are conflicting (Grant et al., 2012), possibly reflecting specific ground and management conditions; one study (Hobbs, 1984) showed that, together with grazing, a short-term 10-year burning rotation was more beneficial to *Sphagnum* than a 20-year rotation.

In an English Nature review of the historical effects of burning and grazing (Shaw et al., 1996), it was stated that "burning is not usually recommended for management of blanket bog although there may be a case for its infrequent use in some circumstances". This review concluded that regular burning regimes are likely to be damaging to the wildlife interests of blanket bog and wet heath, although if carried out sensitively, they could be advantageous to some species of these habitats. Shaw et al. (1996) also concluded that it was impossible to define one optimal management regime to cover all areas of blanket bog and upland wet heath throughout Britain due to local factors, past management regimes, precise management objectives and possible conflicts of

interest. In a more recent English Nature review of burning in the uplands (Tucker, 2003), it was recommended that burning should not be carried out on blanket bog “unless the conservation benefits are clear, or for the creation of firebreaks to prevent wildfire in high risk areas”. Firstly, removal of biomass through burning reduces the input of litter - and thus carbon - into the soil organic matter pool. Secondly, burning can expose the peat surface, leading to increased runoff and erosion of surface peat, especially on slopes, which also results in reduced carbon accumulation. However, burning creates charcoal, which is assumed to be resistant to decomposition, which might reduce any long-term carbon losses (see Clay et al., 2010b).

The literature review agreed with O’Brien et al. (2007) that further investigation of the impacts of the frequency and intensity of prescribed burning is required. Particularly poorly understood are impacts on *Sphagnum* mosses, which are frequently considered to be susceptible to burning, as well as the full effects of rotational burning, and how these effects vary according to differences in burn regimes (Grant et al., 2012). Moreover, there is little experimental evidence on management impacts in relation to slope. In particular, more research is needed to investigate direct and indirect (e.g. via changes in plant species and vegetation cover) effects on dissolved organic carbon (DOC) and particulate organic carbon (POC) through erosion and runoff, particularly considering topography and catchment-scales, and carbon input in the form of charcoal.

2.3 Cutting (mowing)

In circumstances where burning may not be an appropriate management practice, cutting of heather by mechanical means is another way of managing heather (DARDNI, 2011; Ward et al., 1995). Although cutting could be seen as a year-round management tool, heather cutting should not be carried out after 15th April nor throughout the summer months, as ground-nesting birds are likely to be present (SEERAD, 2001).

Cutting is mostly used to encourage heather regeneration and covers a range of mechanical techniques including cutting, swiping and flailing, with machines which use either blades or chains (MacDonald, 1996). However, it may be difficult to manoeuvre larger machinery, particularly on uneven or sloping ground, and forage harvesters (front cutting) are likely to suffer more from damage than flails or swipes (MacDonald, 1996). Where brash is to be left on site, it needs to be chopped very finely and evenly distributed to avoid smothering heather regrowth and other vegetation. A potential advantage of leaving brash after cutting is the reduced actual evapotranspiration from the covered ground compared to exposed burn areas, which limits any negative rain/erosion effects (hence why brash is used for restoration purposes) whilst also returning carbon to the soil, thus aiding peat accumulation.

Cutting might be expected to produce faster initial rates of heather regeneration than burning (Liepert et al., 1993), since a greater number of sprouting centres are retained (Mohamed and Gimingham, 1970) and re-sprouting heather plants grow much more quickly than heather seedlings (MacDonald, 1996). However, the plant’s potential for re-sprouting from dormant buds on the stem base declines as bushes become larger and more woody (Hobbs & Gimingham, 1984), meaning cutting is more likely to be beneficial to vigorously growing heather in the building or early-mature stages (MacDonald, 1996) if the aim is to produce or maintain high heather cover. Therefore, different ages of heather could be cut to assess mowing impacts on heather re-growth in relation to reducing heather cover and supporting ‘active’ peat-forming and ‘typical’ bog vegetation.

As reported by Liepert et al. (1993) and Cotton and Hale (1994), in old stands of heather, cut areas may take longer to regenerate than burnt areas, and sometimes may not regenerate at all (Backshall, 2001). This is because in old stands of heather, regeneration is almost entirely from seedlings whose growth is slower than that from stem regrowth (i.e. Liepert et al., 1993) and this regeneration from seed is worse in cut plots than burnt plots (Liepert et al., 1993; MacDonald, 1996). This may be because, if the brash is left behind after cutting, this can inhibit seed germination and establishment due to smothering (MacDonald, 1996). MacDonald (1996) therefore

argues that cutting is less suitable than burning for restructuring and regenerating old heather stands. This is supported by Liepert et al. (1993) who compared heather regeneration after cutting (with brash removal) and burning on both old and young heather stands in the North York Moors. In contrast, the Moorland Association in a website article in 2012 (since removed) asserted that mature, and especially degenerate, heather will regenerate more quickly after cutting than after being burnt. There is only one study to date comparing the effects of burning to those of cutting with brash being either removed or left on site (Worrall et al., 2013); this single site study in the Peak District (without pre-management change monitoring) revealed lower water tables for areas under burn than those under mown management, with the lowest water tables under intact vegetation (likely reflecting differences in evaporation losses). However, the review highlighted that evidence on these issues in truly replicated studies is lacking and more research is clearly needed.

A distinct advantage of cutting is that it is much less constrained by weather conditions than burning (Tucker, 2003), although access for cutting machinery is easier when conditions are drier and the ground is firmer (MacDonald, 1996). However, not all areas are accessible to large scale mowing, and less so, bailing equipment. The capital costs of cutting are likely considerably higher per unit area (particularly when using a contractor) than burning (MacDonald, 1996). However, if labour and suitable machinery are already available then cutting or swiping can be done for about half the cost of traditional controlled burning (Tucker, 2003). Costs to society and the environment through 'out of control' fires are difficult to measure and have not been included in such calculations. Furthermore, no research was found reporting impacts that may result from the interaction of grazing and cutting.

However, there are some drawbacks to cutting (e.g. Backshall et al., 2001; Tucker, 2003; DARDNI, 2011). There are many areas where the ground is too steep, rocky, or wet for machinery to be used safely or without causing damage to the vegetation and peat. In drier areas, the cut material may not decay quickly, suppressing regeneration of heather and other vegetation if a thick mat of cut material is produced. Cutting machinery can potentially also cause damage to archaeological features. Cutting, like burning, is a drastic event for the vegetation and its associated fauna. The machinery used in cutting can damage fragile peaty ground, reduce complexity of micro-topography (by cutting tops off hummocks and levelling peat surfaces), cause compaction and is restricted to areas with suitable topography. Cutting is not recommended for blanket bog where regeneration is very slow and machinery can damage the vegetation and should not be carried out within 10 m of any watercourse to minimise pollution and excessive trampling of watercourse edges (DARDNI, 2011).

It is important to note that currently, cutting is seen mainly as a tool for achieving optimum heather re-growth and coverage as well as, for example, fire breaks (e.g. Backshall et al., 2001). It could, however, also be used to regulate heather re-growth and encourage peat forming bog vegetation amongst the heather by changing the frequency of mowing or by leaving different quantities of brash, although no published studies could be found assessing the impacts of repeated cutting at high frequency on heather coverage, vegetation composition and blanket bog species biodiversity. Repeated cutting every 5-10 years could give an advantage to some of the peat forming bog species by suppressing heather regeneration and reducing its dominance.

2.4 Herbicide

In theory, herbicides such as Asulam could be used to control heather in the same way as formerly used to control bracken or *Molinia*. However, there are numerous environmental concerns about the use of herbicides to control moorland vegetation. Spraying-related water quality issues (Holden et al., 2008) and ecological implications are of immediate concern in the UK, especially in England, as most blanket bog areas are within major drinking water supply areas and also part of ecologically important areas for biodiversity (i.e. SSSI status).

2.5 Other options

Several other management options such as natural bio-control, repeated ground pressure and liming were briefly discussed in the literature review, but only the addition of *Sphagnum* propagules (as pellets), which were already used in restoration sites across the Peak District, was recommended to be considered as part of the experimental platform. This could specifically target peat-forming species, currently largely absent from the experimental blanket bog sites, which are suitable as indicator species (e.g. for pH, moisture and nutrient levels) for blanket bog function (O'Reilly, 2008).

2.6. Recommendations for experimental design and treatments

In conclusion, the most likely beneficial impact on supporting 'active' blanket bog vegetation, and ensuring peatlands' hydrological function, is to be gained from grip blocking. As large areas of the UK blanket bogs are under burn rotation for grouse management, which are increasingly undergoing grip blocking, it was clearly important to include such representative areas (i.e. ideally with blocked, inactive grips) in this project. Initially, the literature review (**Annex A**) recommended including a grazing exclusion treatment. However, the workshop participants felt this plot-level treatment was too complex within a plot-level assessment and was therefore rejected (**Annex A.1**). Instead, grazing exclusion plots were dropped in favour of 'do nothing' uncut comparison plots to allow for a more balanced experimental design. Importantly, the review found that there has been surprisingly little research carried out on the effects of mowing compared to those of burning particularly on species diversity, water quality, GHG fluxes and carbon balance of peatlands and heather-dominated habitats. The major alternative managements used in the field experiment were therefore burning and mowing. Moreover, as most sites contain some *Sphagnum* moss species, yet might be missing some key peat forming species, the addition of such specific *Sphagnum* re-introductions was recommended to be considered at the plot-scale.

Therefore, based on the available information from the literature review and the project workshop, we designed a paired experimental platform (comparing burning versus mowing) at three sites in Northern England. The experiment served as a rigorous experimental and replicated test-bed, at both the catchment- and plot-scale. Whilst the two major managements were applied at the catchment-scale (burning versus mowing with leaving brash), seven treatments were investigated at the plot-scale (5 x 5 m) within the respective sub-catchments:

- Burning of heather (catchment-scale)
- Burning of heather and *Sphagnum* addition
- Mowing with brash left (catchment-scale)
- Mowing with brash left and *Sphagnum* addition
- Mowing with brash removed at plot level only
- Mowing with brash removed and *Sphagnum* addition
- Uncut comparison areas within the mown catchments

The unique strength of the experimental approach was therefore the combination of catchment- and plot-level replication. However, the uncut and brash removal options could only be assessed at the plot level, lacking a catchment approach, which is particularly relevant for any hydrological and flow impact assessment.

The paired catchment-scale comparison at each site of a 'business as usual' burn rotation versus an alternative mowing management was recommended at the usual management patch size (~0.3 ha), with multiple patches aimed to achieve with each management intervention a total of about 20% of the catchment area (equal to about 25% of the total heather dominated area) within each catchment every two years (equal to a ~10 year rotation).

This would enable management of the heather dominated areas within the entire catchment within a 10 year period. Crucially, mowing areas were of similar patch size and total area proportionally to that of the paired burn catchment. The catchment scale approach allowed several important knowledge gaps identified in the literature review to be addressed, particularly in relation to landscape processes of runoff, erosion and stream flow and resulting carbon export. The uncut plots allowed the effects of management to be placed in context of effects of other environmental factors (e.g. climate) regardless of management as well as addressing the fundamental question of long-term development of an unmanaged heather stand (e.g. the effect of opening up of the increasingly rank heather on the bog vegetation community). Moreover, the uncut plots (or other areas of old heather) offer the possibility of a later assessment of heather re-growth in relation to natural layering (Rawes & Hobbs, 1979) and/or stand age (i.e. re-growth after mowing on older heather in comparison of the 15-20 year old heather from the initial experimental mown plots). The *Sphagnum* addition plots also offered a possibility to facilitate a later management addition such as increased mowing frequency (i.e. compared to the paired mown plots without *Sphagnum* addition) or further *Sphagnum* addition (as plugs) on burnt plots as part of the effectiveness and potential of restoration burning.

Notably, the experimental approach followed O'Brien et al.'s (2007) recommendation for long-term monitoring aimed at unravelling management impacts on blanket bogs to promote recovery and return to favourable, 'active' conditions. Clearly, timescales are important and although some changes can be expected within the current 5-year project period, a continuation beyond five years was recommended by the project team following the literature review and by the stakeholder workshop participants to assess transient effects versus more permanent, and thus policy relevant, shifts in plant community responses and associated long-term impacts on ecosystem services. Crucially, the catchment-scale replication allowed a range of climatic and habitat site conditions to be included and a more meaningful interpretation of management impacts by (i) ensuring the same methods were applied throughout and (ii) considering a broader range of climatic and site conditions. The lack of coherent landscape scale replication in previous studies clearly limits the value of study outcomes across other areas (i.e. every study and site is different). Finally, a pre-management change period was recommended to allow a Before After Control Impact (BACI) assessment, which enables interpreting findings in light of existing pre-management change differences (i.e. differences unrelated to the management change). To our knowledge, BD5104 offers the only current UK, and in fact global, long-term replicated plot- and catchment-scale experiment on blanket bog to provide such a robust test platform, particularly in relation to the most recently identified knowledge gaps and recommendations for future research (i.e. Harper et al., 2018).

3. Overview of field sites and experimental design

3.1 Field sites

The three study sites were all located in north-west England (**Figure 1**). The names used to identify the sites throughout this report are **Nidderdale**, **Mossdale** and **Whitendale**. Each site offered two adjacent (Nidderdale and Mossdale) or closely located (Whitendale) sub-catchments of similar size (~10 ha), with each being allocated either burning or mowing management after an initial pre-treatment period. Each sub-catchment had one main stream (see **Figure 1**), and their proximity allowed all to be reached within one day when necessary.

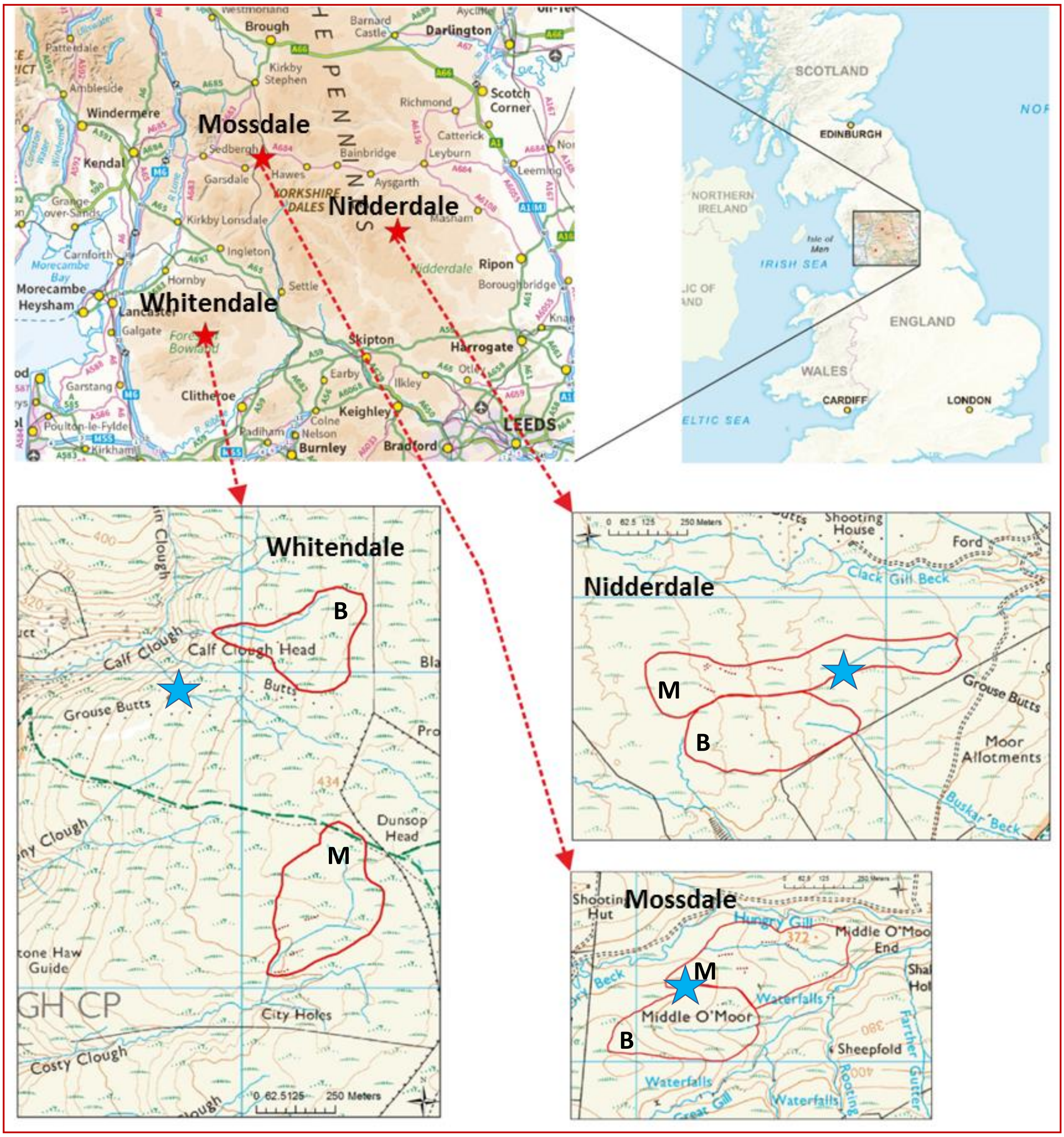


Fig. 1 Location of the three study sites in north-west England (**top maps**, red stars). The catchment boundaries (thick red lines) with the burnt (B) and mown (M) catchments and automated weather station (blue star) are detailed in the **lower maps** for Whitendale, Mossdale and Nidderdale. Source: MiniScale® [TIFF geospatial data], Scale 1:1000000, Tiles: GB, Updated: 3rd December 2015, Ordnance Survey (GB), Using: EDINA Digimap Ordnance Survey Service, http://digimap.edina.ac.uk, Downloaded: 2016-09-09 14:35:01.73. Note the main stream within each sub-catchment.

3.2 Site characteristics

The sites were chosen based on a set of key criteria: all were classed as blanket bog with a mean peat depth of over 1 m, and were managed as grouse moors. Typically, the sites were managed with a 10-15 year burn rotation (based on gamekeeper information) and all had a long history of burning (more than 100 years; based on estate information). All sites had more than 50% *Calluna* cover, with at least some existing bog vegetation in the form of *Eriophorum* and *Sphagnum* species, and had a low stocking density of <0.5 ewes ha^{-1}. Whilst two of the sites had some old and mostly infilled grips at low density, the third site without grips included some natural gullies. The sites allowed for a pair of similar sized (~10 ha) sub-catchments, each with a main stream (**Figure 1**), and their proximity allowed reaching all within a day when necessary.

Nidderdale is located on the Middlesmoor estate in upper Nidderdale, which lies within the Yorkshire Dales National Park, UK, at 54°10’07”N; 1°55’02”W (UK Grid Ref SE055747) about 450 m a.s.l. The site had an average annual air temperature of 7.2°C and annual precipitation of 1587 mm during the five year study period. It is situated on mudstone from the Millstone Grit group with intermingled sandstone (Kidd et al., 2007). The soil is a poorly draining organic peat (Winter Hill series) with an average depth of 1.6 m ± standard deviation (STDEV) of 0.3 m across the experimental plots and peat depth across the catchments ranged from 0.2 m to 2.9 m (according to manual peat rod measurements made in 2012). Most of the grips within the study area, which were dug about 40 years ago, were naturally infilled by 2010 and no further grip blocking took place during the study period. The two sub-catchments were ~11 ha (burnt) and ~13 ha (mown).

Mossdale is located in Upper Wensleydale within the Yorkshire Dales National Park at 54°19’01”N; 2°17’18”W (UK Grid Ref SD813913) about 390 m a.s.l. During the five year study period, the average annual air temperature was 7.2°C and annual precipitation was 2029 mm. The site lies on limestone which is overlain by thin sandstone and covered with mudstone (Hall, 1979). The soil is a poorly draining organic peat (Winter Hill series) with an average peat depth of 1.2 ± 0.4 m at the experimental plots and peat depth across the catchments ranged from 0.3 m to 2.1 m (manual peat rod measurements made in 2012). Most of the grips within the study area, which were dug about 40 years ago, were naturally infilled by 2010. Three minor ditches (already naturally infilled and running along the contours) were unintentionally blocked (by restoration work) during the study period on the control catchment on 20th November 2013. The two sub-catchments were ~8 ha (burnt) and ~10 ha (mown).

Whitendale is located within the Forest of Bowland (an Area of Outstanding Natural Beauty; AONB), Lancashire, at 53°59’04”N; 2°30’03”W (UK Grid Ref SD672543) about 410 m a.s.l. The average annual air temperature was 7.6°C and annual precipitation was 1858 mm during the five year study period. The site is situated on interbedded sandstone and mudstone with areas solely of mudstone (Ewen et al., 2015). The soil is a poorly draining organic peat in the Winter Hill series (Ewen et al., 2015) with an average peat depth of 1.7 ± 0.4 m at the experimental plots and peat depth across the entire catchment area ranged from 0.2 m to 4.5 m (manual peat rod measurements made in 2012). This study area had no grips, although gullies (similar to grips but naturally formed) were present in both catchments. The two sub-catchments were ~8 ha (burnt) and ~11 ha (mown).

All sites are heather dominated blanket bogs under intense grouse moor management, offering fairly flat areas; Nidderdale, Mossdale and Whitendale had an average slope on the experimental plots of 4º; 6º; 8º (± 3º STDEV), respectively. Each site offered two adjacent (Nidderdale and Mossdale) or closely located (Whitendale) sub-catchments of similar size (~10 ha), with each being allocated either burning or mowing management after an initial pre-treatment period. Recent aerial pictures (Google Earth; **Figure 2a** and **2b**) offer a basic management comparison. Whereas rotational burning patches were clearly visible at Mossdale and, in particular, at Nidderdale, burning activity was less obvious at Whitendale (**Figure 2b**). Recently (2009) mown strips were apparent in both sub-catchments at Whitendale, but ground truthing revealed an intense burn rotation amongst the mown strips, which were initiated by the local water company (United Utilities) as a trial without monitoring.

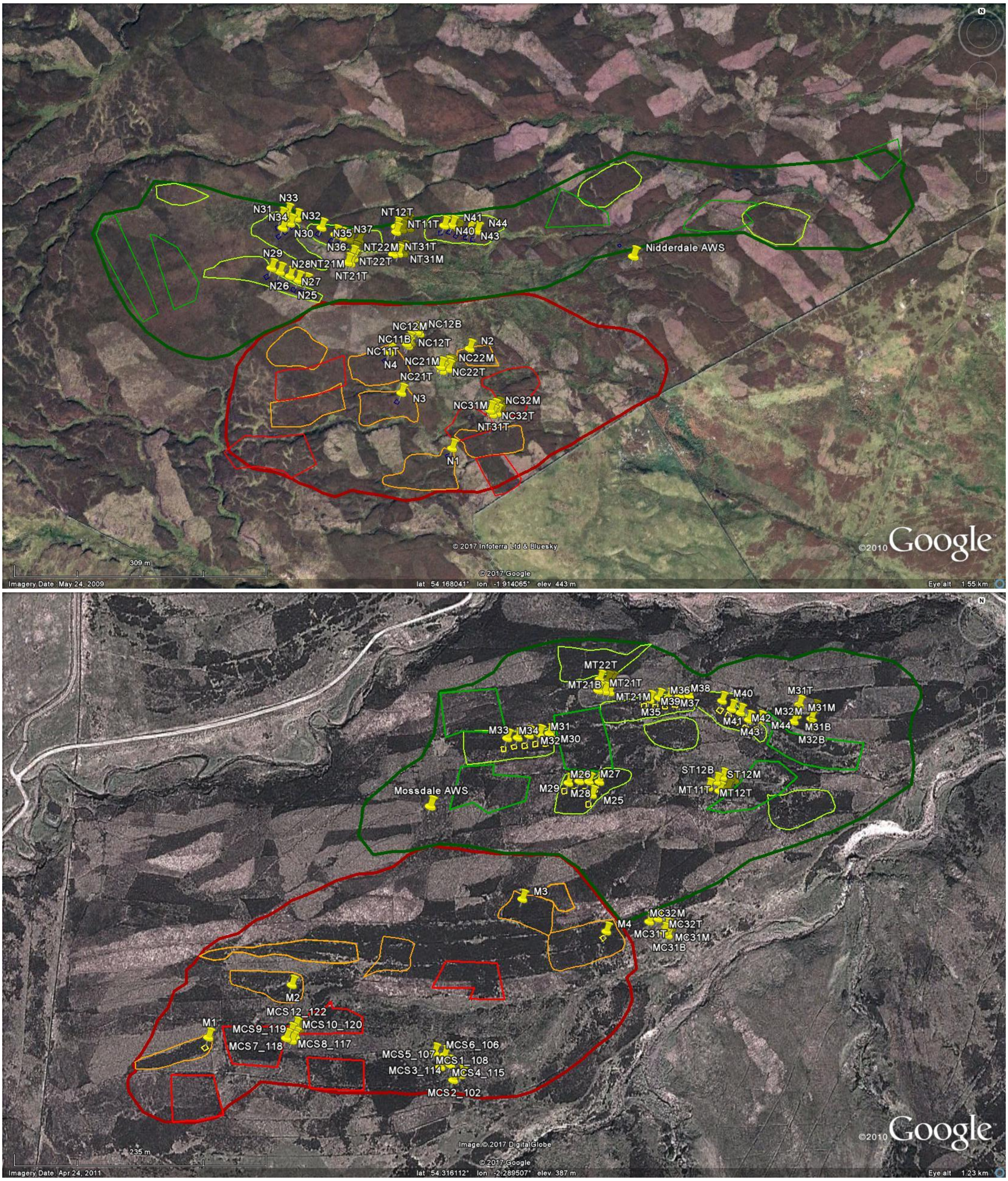


Fig. 2a Outlines for the paired sub-catchments (red = burnt; green = mown) at the two sites: Nidderdale (**top**) and Mossdale (**bottom**) with pictures obtained from Google Earth (2010). Shown are the GIS layers for the large catchment boundaries, the plot (5x5 m) and slope locations (yellow pins) and the burnt and mown areas (small polygons) with orange and pale green polygons for 2013 and red and dark green polygons for 2015 management interventions, respectively. All sites show active grouse moor management (burnt strips). Scale bars (i.e. entire white marker length shown in the bottom left of each picture) are 309 m for Nidderdale and 235 m for Mossdale.

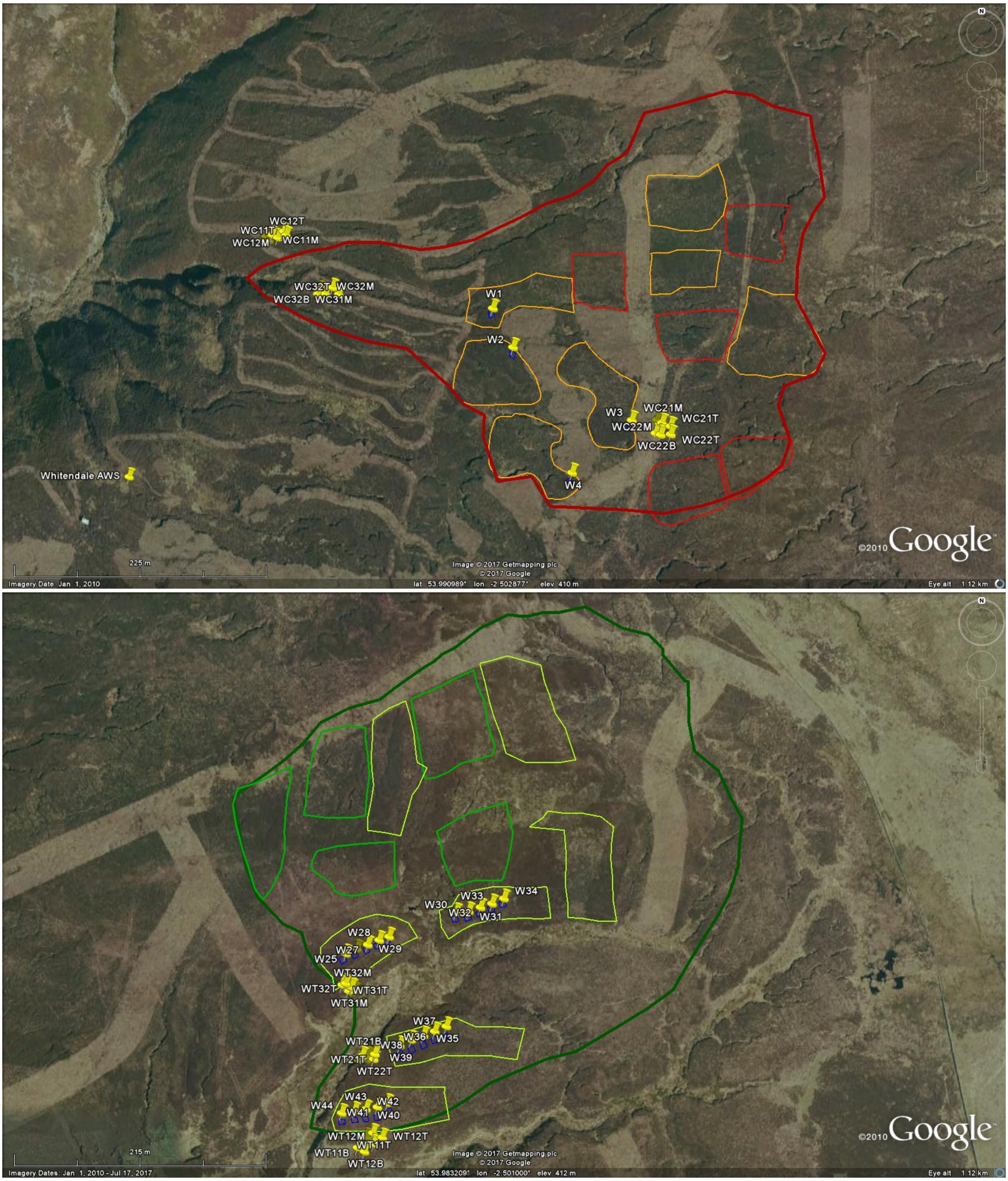


Fig. 2b Outlines for the paired sub-catchments (red = burnt; green = mown) at the Whitendale site: burnt sub-catchment (**top**) and mown sub-catchment (**bottom**) with pictures obtained from Google Earth (2010). Shown are the GIS layers for the large catchment boundaries, the plot (5x5 m) and slope locations (yellow pins) and the burnt and mown areas (small polygons) with orange and pale green polygons for 2013 and red and dark green polygons for 2015 management interventions, respectively. Both sub-catchments show some active grouse moor management (burnt areas) and also previously (2009) mown areas (stripes). Scale bars (entire white marker length shown in the bottom left of each picture) are 225 m for the burnt and 215 m for the mown catchment.

There were further noticeable differences in habitat conditions. Ground truthing revealed erosion on previously burnt areas (**Figure 3**), particularly on steeper slopes (~15-25^{o}) at all sites, and at Nidderdale most of the lower

lying areas had shallow peat with some rocks visible. However, mowing was suitable on all mown sub-catchments thus allowing comparisons with any burning-associated bare peat and erosion aspects (**Figure 3**).

Fig. 3 Examples of site conditions **from left to right**: a) eroding bare peat areas on past burns at Nidderdale; b) rocks showing through shallow peat on past burns at Nidderdale; c) dense brash and moss cover on mown areas at Mossdale three years after mowing; d) ~15^{0} slopes at Mossdale showing considerable erosion and cracking on a past burn.

3.3 Experimental design

For each site, two similar adjacent sub-catchments were randomly allocated either a burning or mowing management at the catchment scale, with various plot-level managements, including an additional uncut plot-level management within each mown sub-catchment. The entire manipulative experiment was based on a Before-After-Control-Impact (BACI) design (Schwarz, 2015; Stewart-Oaten et al., 1986), to enable robust statistical analysis of the effects, and therefore included almost a full year of pre-management monitoring. Each site was visited in November 2011 to assess site conditions and determine suitability for the project; an appropriate central point for a weather station (**Figure 4**) was determined between two the sub-catchments, which were of similar size and manifested similar conditions (e.g. vegetation, management, slope, peat depth).

Fig. 4 Argocats enabled accessing remote areas during setup (**left**), away from public footpaths. Weather stations (**middle**) were protected from sheep by a fence with reflective plates to be visible to birds. Individual 5x5 m plots (**right**) were marked by low wooden posts, with uncut plots marked by larger fence posts to ensure about 1 m of unmown edge around the plot.

The rigorous experimental design enabled a robust statistical approach, addressing the need for a pre-treatment period (i.e. before a change in management) and providing replication at plot and catchment scales, which is both scientifically rigorous and relevant to practitioners. The paired catchment design allowed for comparison of the main managements (burning and mowing) across sites whilst the plot-level treatments within the sub-catchments could also be compared at each site. This design is unique within a peatland context; it aims to set a precedent for future ecological work and offers a potential long-term research platform of national and international significance.

Sub-catchment boundaries for the three sites were defined based on the watershed, first by defining the rough outline of each catchment using contour lines on a detailed map, and then walking the top of the identified ridge around each sub-catchment using a GPS to accurately record the outline. A V-notch flow weir (with a notch angle of 90°) was constructed from durable PVC at the outflow of each sub-catchment (see Section 4.2.8 and Appendix 4). One sub-catchment of each site was randomly assigned to a business-as-usual burning management with the other being assigned to management by mowing. Within each sub-catchment four blocks each with one plot-level replicate**,** were defined with at least 50 m between blocks (e.g. **Figure 5**). In the burning sub-catchment, each block contained two plots; **FI** plots were solely burnt and **FI+Sp** plots were burnt with *Sphagnum* propagules added (see Section 4.2.6.2). In the mown catchment, each block contained five randomly allocated treatment plots; **LB** plots were mown with the brash left, **BR** plots were mown with the brash removed (see **Figure 9**), **LB+Sp** plots were mown with the brash left and *Sphagnum* propagules added, **BR+Sp** plots were mown with brash removed and *Sphagnum* propagules added and **DN** plots were left uncut as the 'do nothing' control.

All **plots were 5 x 5 m**, with a 5 m gap between each plot, and were marked out with wooden corner posts protruding approximately 50 cm from the peat surface. In the lowest corner (flow direction) of each plot (see **Figure 5** for a schematic diagram of a typical plot), a temperature logger (Tinytag Plus 2 – TGP-4017 data logger, Gemini Data Loggers Ltd, Chichester, UK) was placed on the peat surface and covered by a reflective lid secured by plastic mesh and pegs. Another logger unit (TGP-4520) measured the temperature at the peat surface (with the probe covered by 0.5 cm of peat to prevent any direct sunlight heating the sensor) and at 5 cm depth on FI, LB and DN plots only. Above the temperature logger lid, a WTD meter (see Section 4.2.7) was inserted into a 1 m deep hole cored in the peat and a peat rod (steel; 12 mm diameter) was inserted into the bedrock (to enable measurement of peat surface changes). Additionally, each plot contained a Rhizon sampler (type MOM 10c, 2.5 mm diameter, van Walt, UK) extending to 10 cm peat depth which allowed for periodical sampling of peat pore water (see Section 4.2.10.1). The instruments were covered by a stainless steel mesh cage which was pegged at the bottom and folded at the top to prevent sheep damage whilst allowing easy access. Circular flux areas for repeated methane (CH_4) and total CO_2 soil respiration (SR), root-free SR (SR_c) and net ecosystem exchange (NEE) measurements were chosen along one side of the plot and marked with metal pegs. A **1 x 1 m sub-plot** was marked in each plot in a different corner to that of the WTD meter for detailed vegetation monitoring (**Figure 5**).

Each sub-catchment contained a further three experimental slope locations between the four main blocks containing the plot-level replicates to capture a range of aspect and slope conditions across the sub-catchments. Each slope location consisted of six plots, set out in two rows down a slope, with each plot containing a WTD meter (measuring to 50 cm depth) and surface temperature loggers and Rhizon samplers (as above). Mean slopes on slope plot locations on Nidderdale, Mossdale and Whitendale were 8^o, 10^o and $14^o \pm 4^o$ STDEV, respectively.

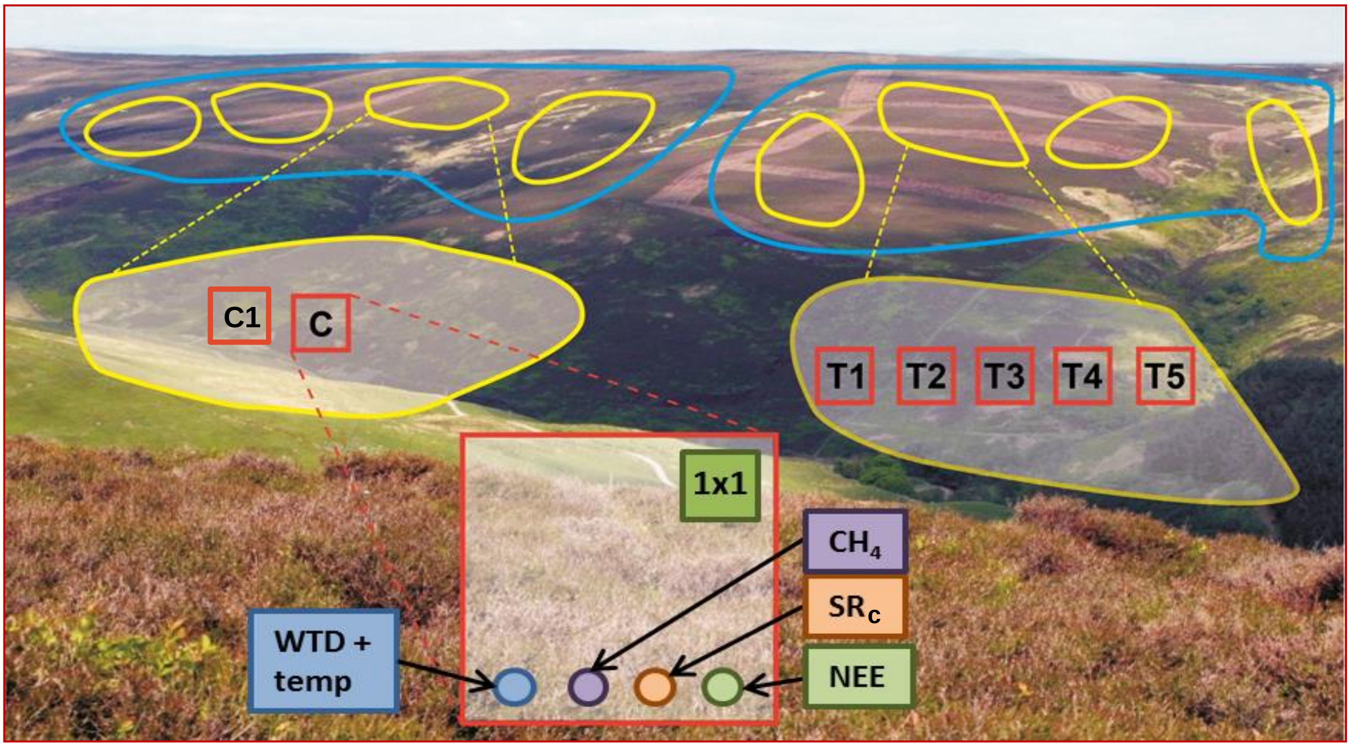


Fig. 5 Typical site layout (i.e. a schematic only and not one of the actual project sites) of the two sub-catchments (blue outlines) with four blocks (yellow outlines) each. Each plot (red outlines) is 5x5 m. Control (C) plots were burnt (FI) and the additional C1 plots were burnt with *Sphagnum* propagules subsequently added (FI+Sp). Treatment (T1-T5; randomly allocated) plots in the mown sub-catchment were either mown with brash left (LB) or brash removed (BR), were mown with *Sphagnum* propagules added (LB+Sp; BR+Sp), or were left uncut as 'do nothing' comparisons (DN). Each plot contained a corner 1x1 m area (green square) for detailed vegetation monitoring, a circle for CH_4 and total soil respiration (purple circle), root-free soil respiration (SR_c) areas for decomposition (brown circle), and NEE flux (green circle) measurements and a mesh cage with a dipwell and temperature logger (blue circle). Each sub-catchment also had three slope locations, one positioned between each of the four main blocks (i.e. not shown but located between the yellow outlines).

Management change started with the first management phase in 2013 (see **Figure 2** for orange and pale green polygon outlines), burning (Nidderdale: 5th March; Mossdale: 1st March; Whitendale: 21st February) and mowing (Nidderdale: 11th April; Mossdale: 9th April; Whitendale: 7th March) on all blocks and on three additional areas per sub-catchment (~0.24 ha each). In the second management phase in 2015 (see **Figure 2** for red and dark green polygon outlines), five new areas (~0.25 ha each) within the sub-catchments were burnt (Nidderdale: 10th and 14th April; Mossdale: 19th March; Whitendale: 18th March) and mown (Nidderdale: 13th January; Mossdale: 31st March; Whitendale: 13th and 14th March). Burning was only partially successful at Whitendale in 2013 (due to patchy snow cover), leading to protection of the moss and vegetation layer.

Burning was done manually with gas torches setting alight the outside of the targeted heather area and controlled burning of the area inside controlled by the direction of the wind and beating around the edges with fire beating shovels or rubber flap fire beaters, whilst mowing was done using adapted tractors with double wheels (which at Mossdale were also fitted with tracks to the back wheels) to reduce ground pressure to about 2 pounds per square inch (psi). At Nidderdale a more basic and lighter mowing machinery (a small Case International 4230 tractor, 85 horse power, with a back-fitted simple Bomford Topper (RS18) flail) was used, compared to the heavy machinery performing double-chop mowing at Mossdale (a New Holland, 120 horse power, with back-fitted Major hammer flails) and Whitendale (a Deutz Fahr tractor, 150 horse power, with a back-fitted heavy duty Ryetec flail mower).

3.4 Weather stations, catchments and management

Although the three sites had the same basic setup, landscape features required placing the weather stations in suitable locations within the overall catchment, which coincided with different vegetation and topography. Moreover, although very closely located to each other, the paired catchments also varied slightly in vegetation and topography. Finally, individual site management achieved similar burning and mowing results. The pictures below (**Figures 6-9**) show the main characteristics of the three sites, their catchments and management practice.

Fig. 6 The automated weather stations (AWS) at (**from left to right**) Nidderdale, Mossdale and Whitendale. All sites are on heather dominated grouse moors but vegetation at the Mossdale AWS location is shown re-growing after a recent burn. Pictures were taken in December 2016.

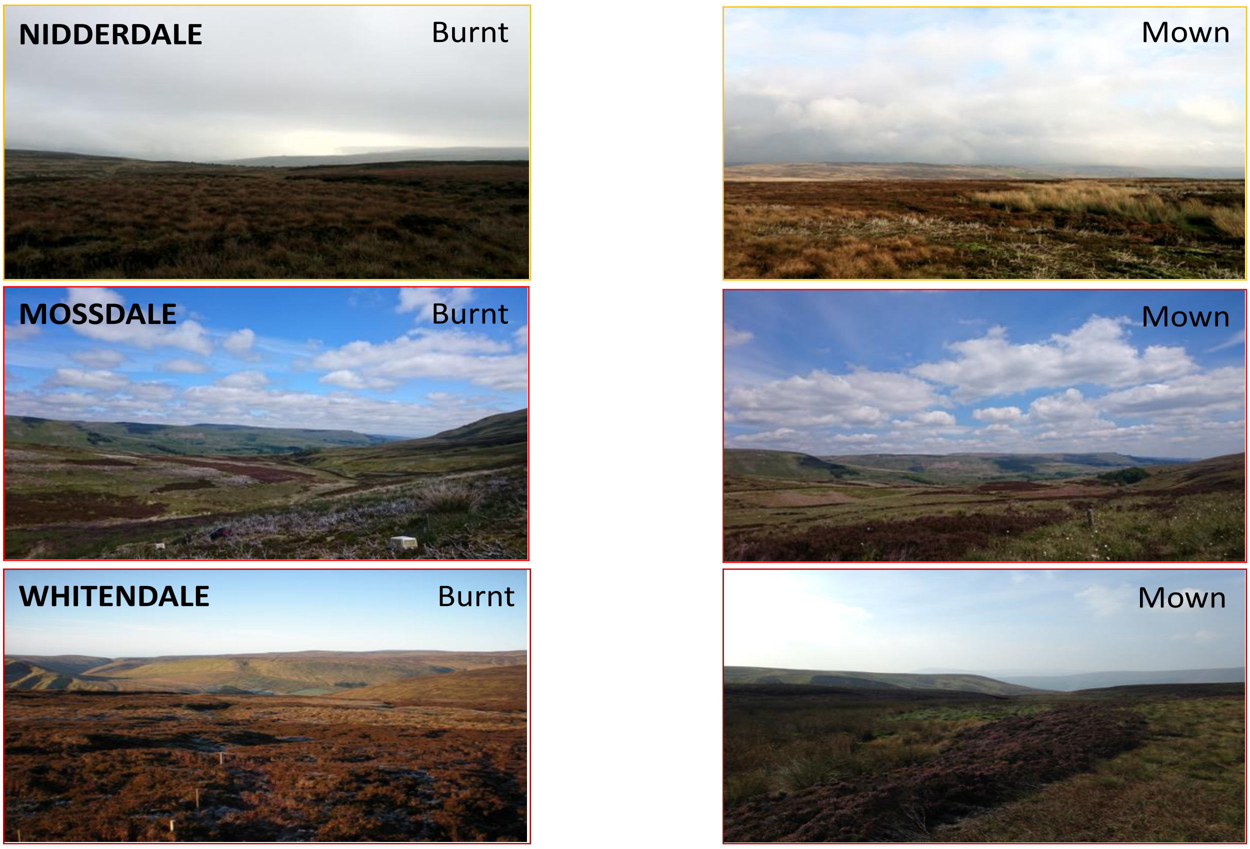


Fig. 7 The six sub-catchments shown from an elevated point. Shown are the pairs of burnt (**left**) and mown (**right**) sub-catchments for Nidderdale (**top**), Mossdale (**middle**) and Whitendale (**bottom**). Pictures were taken in September 2015.

Fig. 8 The six sub-catchments (**from left to right**: Nidderdale, Mossdale, Whitendale) shown during burning (**top row**) and after mowing (**bottom row**) in March/April 2013. Also note the air pollution (visible as smoke) impact during the burning.

Fig. 9 The three mowing arrangements (**from left to right**): Nidderdale, Mossdale and Whitendale. On average, vegetation was mown about 12 cm above the peat surface and the heather brash returned to the surface was about 5-10 cm long, with the coarsest brash at Nidderdale and the finest at Whitendale. The initial brash layer after mowing was around 5 cm thick. Brash was removed from mown (BR) and mown with added *Sphagnum* (BRSp) plots by manual raking (~4-5 times ca. 50 L brash were collected from the 5x5 m plots in 70 L bags, see picture on the **far right** and deposited in adjacent areas.

3.5 Overview of measurement methods

Collection of baseline climatic data with weather stations started at all sites in March 2012, as did the continuous monitoring of water table depths and soil temperatures at the plot level. Flow weirs were installed in 2012 on 10th July at Nidderdale, 11th July at Mossdale and 12th July at Whitendale. Water sampling from the flow weirs commenced thereafter and samples were taken monthly, resulting in 9 pre- and 46 post-management samples. Carbon fluxes (NEE, SR and CH_4) were measured 3-4 times per year starting in July 2012 and resulting in 3 pre- and 15 post-management measurements. Roots were cut around the SR_c plots twice a year (generally in spring and autumn), starting after the first flux measurement, to provide decomposition only fluxes (i.e. without root respiration) to compare with total SR (as measured on the CH_4 plots with root respiration). Soil pore water samples were taken 3-4 times per year from March 2012 onwards, providing 3 pre- and 11 post-management samples. Vegetation surveys on the 5 x 5 m and 1 x 1 m plots were performed annually in September or October. Cranefly numbers were counted in 2014, 2015 and 2016, with monitoring commencing in April and continuing until the end of July, producing 4-5 survey dates per year. Ground Penetrating Radar (GPR) surveys of peat depths and pipes were conducted in August 2012 and again in late March/early April 2016. The first survey included a continuous (GPR drawn on a sledge) in addition to a manual (walking) survey. Peat depth was measured twice at each plot, once before (29/08/12) and once after (25/03/14) management. Peat surface fluctuations (level) were assessed during 2014 and 2015 and micro-topography for each plot was captured during 14-16th September 2015.

3.6 Data collation, project data sets, quality control and sample bias

All project data have been collected and analysed using protocols that conform to Defra's and NERC's JCoP (Joint Code of Practice) standards. The protocols have been written as standard operating procedures (SOP). Project data are stored at York and are backed up securely. Various quality control measures were applied, depending on the data and the analyses. Outliers for water quality parameters (e.g. very high DOC) were identified using Z-scores and rejected (threshold set to 95% confidence), while threshold R^2 values were used to reject flux data (e.g. very low and unstable (R^2 <0.4) CH_4 fluxes were set to zero). Moreover, laboratory based analyses followed strict controls, using regularly calibrated instruments (e.g. LiquiTOC, Elementar) and probes (e.g. pH-meter) together with certified standards (e.g. glutamic acid and birch shoot material for C/N analyses).

In general, data loggers performed well but some gaps and offsets needed to be corrected. For AWS data, some battery failures (Whitendale: January 2016) and sensor downtime required cross calibration with other AWS data to fill the short infrequent gaps (there were only six such periods requiring calibration, only one of which was up to three weeks long). United Utilities kindly provided their monthly and daily rainfall totals for Whitendale when the rainfall gauge became blocked in July 2016, which allowed very accurate data gap filling. Soil moisture data (measured using a HH2 meter connected to a SM200 probe, Delta T Devices, Cambridge, England) were calibrated against the results from a laboratory test, which involved drying and weighing intact peat (from Mossdale) at a full range of moisture contents whilst measuring soil moisture with the field probe (see Appendix 1). One flow weir (outflowing the burnt sub-catchment) developed a small leak at Mossdale during part of November 2013; data were gap filled with a robust regression approach using rainfall and flow rates from the mown sub-catchment (for the post-management period). On three occasions, one of the two flow weir WTD loggers failed, but data from the other backup unit was available to gap-fill data. Moreover, regular manual flow height data allowed correcting for any offsets during the sampling period caused by subsidence due to movement. Plot water table data required an offset correction, due to peat surface offsets but also due to inadequate automatic temperature correction of the Omnilog units (which cannot be assessed as data processing is done by the software independently from the user). However, corrections appeared stable over time and were supported by regular manual WTD measurements at all dipwells (five per year). Nine soil temperature units became water damaged and were replaced; missing data were gap filled based on regressions with similar treatments during periods of intact data.

Additional environmental observations were recorded which could help explain observed differences in monitored parameters. For example, heather beetle numbers and associated plant damage were recorded in the summer of 2015, with particularly high numbers at Mossdale and fewer at Whitendale. Moreover, heather plants suffered noticeable frost desiccation at Whitendale, and some damage at Mossdale, due to lack of snow cover in the very cold winter of 2012/13.

In general, data were acquired in a particular order but sometimes switching the plot order and catchments. Soil pore water samples (60 per site) were collected over a two-day period; on the first day, syringes were connected to the Rhizon samplers (usually in the morning at Nidderdale, around midday at Mossdale and in the afternoon at Whitendale) and water was allowed to accumulate overnight (with samples being sheltered from direct sunlight and overnight frost by the temperature logger covers), and samples were collected the next day in reverse site order. All carbon fluxes (24 plots per site) were measured on a single day for each site (and mostly the three sites were visited over three consecutive days), with CH_4 and NEE flux measurements occurring concurrently (between about 10:00 and 14:00), followed by soil respiration measurements (between about 14:00 and 16:00). Any sample order bias was addressed by using the environmental parameters recorded during flux measurements to obtain models explaining the observed fluxes and then upscaling those models to obtain annual fluxes using the sites' automated weather station (AWS) climate data. Flow samples were collected either in a single day from all sites or, if scheduled that month, during the three days of flux measurements, or two day pore water sampling.

4. Summary of results

4.1 Climatic conditions during the study period

In March 2012, automated weather stations (AWS; MiniMet AWS, Skye Instruments Ltd, Llandrindod Wells, UK) were erected between the two sub-catchments at each site (see blue stars in **Figure 1**). The AWS recorded hourly values at 2 m above ground level for wind speed and direction, total radiation and photosynthetically active radiation (PAR), at 1 m above ground level for air temperature, air pressure and relative humidity, at ground level for precipitation, and 8 cm below ground for soil temperature (see Appendix 1 for more information on environmental monitoring). **Table 1** summarizes the average annual site conditions throughout the study period.

Table 1 Summary table showing the annual totals for photosynthetically active radiation (PAR) and rainfall, and the average air (T air) and soil (T soil) temperatures during the study period (2012 - 2016) for the three study sites (Nidderdale, Mossdale, Whitendale). Data for January to March 2012 were estimated based on available regional UK Met Office data and additional climate data for Whitendale and Moor House Natural Nature Reserve (NNR) (see Appendix 1).

Nidderdale	Light PAR(mol m-2)	Rainfall (mm)	T air (°C)	T soil (°C)
2012	5989	2012	6.6	6.9
2013	6858	1311	6.7	7.4
2014	6439	1512	7.9	8.8
2015	6966	1768	7.3	7.7
2016	6664	1333	7.3	8.0
Average	**6583**	**1587**	**7.2**	**7.7**

Mossdale	Light (PARmol m-2)	Rainfall (mm)	T air (°C)	T soil (°C)
2012	5837	2405	6.7	7.2
2013	6581	1699	6.9	7.9
2014	6205	1933	8.0	8.8
2015	6529	2424	7.2	7.9
2016	6576	1684	7.3	8.3
Average	**6346**	**2029**	**7.2**	**8.0**

Whitendale	Light PAR(mol m-2)	Rainfall (mm)	T air (°C)	T soil (°C)
2012	5793	2209	7.2	7.7
2013	6608	1393	7.2	7.7
2014	6126	1714	8.3	8.9
2015	6334	2136	7.5	7.9
2016	6568	1839	7.8	8.6
Average	**6286**	**1858**	**7.6**	**8.2**

Overall, climatic conditions varied between both sites and years (**Table 1**). Mean annual precipitation (MAP) was considerably higher at Mossdale (2029 mm) than at Whitendale (1858 mm) or Nidderdale (1587 mm), whereas both the mean annual air and soil temperatures were highest at Whitendale (7.6 ^oC and 8.2^oC, respectively) and lowest at Nidderdale (7.2^oC and 7.7^oC, respectively). The opposite was the case for total annual PAR, with Nidderdale having the highest average PAR (6583 mol m^{-2}) and Whitendale the lowest (6286 mol m^{-2}). Strikingly, over the five year period both air and soil temperatures increased, by 0.6 ^oC and 1.0^oC, respectively, with a notably warm year in 2014. Across all years, 2012 and 2015 were the wettest, with particularly high rainfall during summer and autumn of 2012, and winter of 2015 (**Figure 10**), while PAR levels were lower in 2012 and 2014.

There were noticeable seasonal differences between years as shown in the monthly climatic data (**Figure 10**). Particularly strong differences were noticed in the onset of spring warming and in rainfall amounts. For example, 2013 showed a very late onset of warming after a very cold winter, while 2015 had fairly low rainfall during spring and summer months, but remarkable high rainfall in late autumn and winter. Moreover, apart from March, 2012 was generally wetter than all other years. PAR generally increased and peaked before temperatures, with a noticeably early peak in the springs of 2012, 2015 and 2016. Overall, highest temperatures were recorded around July, whilst peak radiation occurred around June and greatest rainfall was observed in December.

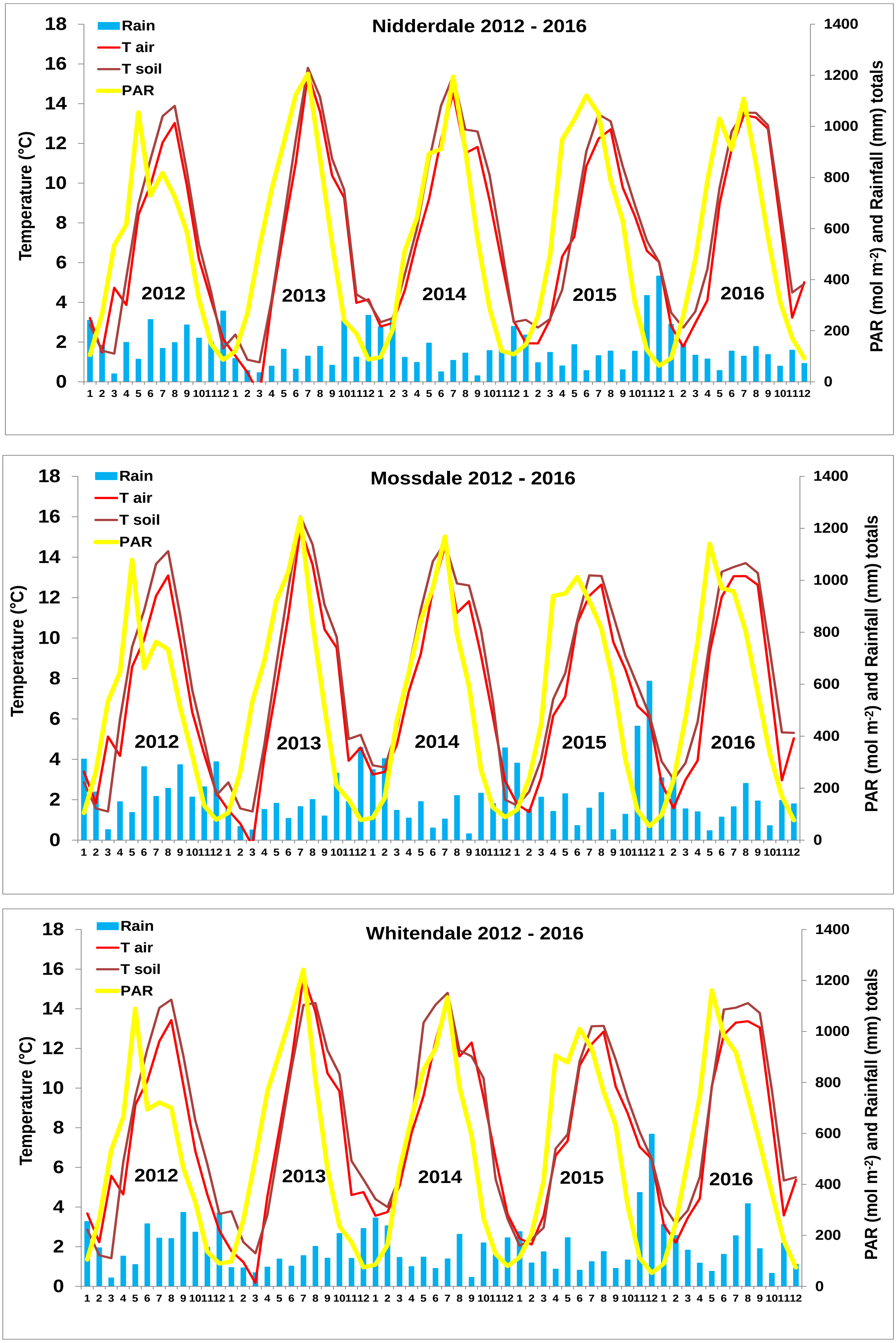


Fig. 10 Summary graphs showing the monthly totals for photosynthetically active radiation (PAR) and rainfall (shown on the same axis but with different units), and the average air (T air) and soil (T soil) temperatures during the study period (2012 - 2016) for the three study sites (Nidderdale, Mossdale, Whitendale). Data for January to March 2012 were estimated based on available regional UK Met Office data and additional climate data for Whitendale and Moor House NNR (see Appendix 1).

4.2 Main monitoring and experimental results

4.2.1 *Manual peat depth surveys*

Peat depth was measured manually next to the water table depth (WTD) meters on all plot and slope monitoring locations across each site, and at random locations across the sub-catchments, in July/August 2012. All locations were surveyed using commercial (Clarke CHT640) 1.5 cm diameter PVC drainage rods (92 cm extendable sections with screw fittings) and peat depth was determined by detecting the sudden increase in force needed to penetrate the peat (i.e. hitting the bedrock/clay layer). Actual depth at all plots was also measured as part of the peat carbon stock assessment (Section 4.2.2) using a custom-made three-sided 5 cm square box corer of 1.1 m length with a cutting blade (avoiding compaction); an extendable (to 5 m) Russian D-corer (van Walt) was used for peat depth below 1.1 m, allowing visual assessment of peat depth and bedrock interface (see **Figure 11** for pictures of the measurement devices and Appendix 2 for more information).

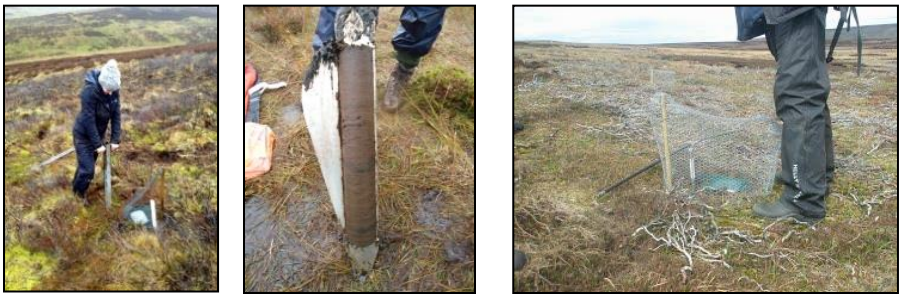


Fig. 11 Manual peat depth surveys used a three sided 1 m box corer (**left**, with the cutting blade in the ground) and a Russian D-corer (**middle**) allowing coring of up to 5 m depth together with manual peat depth rods (flexible and extendable PVC drainage rods) at each plot location (**right**) and randomly throughout the catchment.

Before the onset of management, average maximum peat depth (**Figure 12**) across all sites was similar overall, although it was significantly (Analysis of variance (ANOVA); p< 0.001, n = 24 per site) lower at Mossdale than at Nidderdale and Whitendale (mean ± standard deviation (STDEV): 121±44 cm, 156±33 cm and 168±41 cm, respectively). These values are slightly lower to those measured at Moor House (151 to 231 cm) on similar slope and blanket bog vegetation; this difference may be due to the lower altitude in this study compared to that of the most intensely studied UK blanket bog site at Moor House NNR (about 400 m a.s.l *cf*. 600 m a.s.l., respectively; Garnett et al., 2001 shown in Heinemeyer et al., 2010). Although peat depth in the 'to be mown' sub-catchments differed significantly between sites (p = 0.003, n = 20 per site; again lowest at Mossdale), overall there was no significant difference between sub-catchments within sites (p > 0.05, n = 4 for 'to be burnt' and 20 for 'to be mown' sub-catchment). All sites showed a strong decline in peat depth with increasing slope (see Appendix 2).

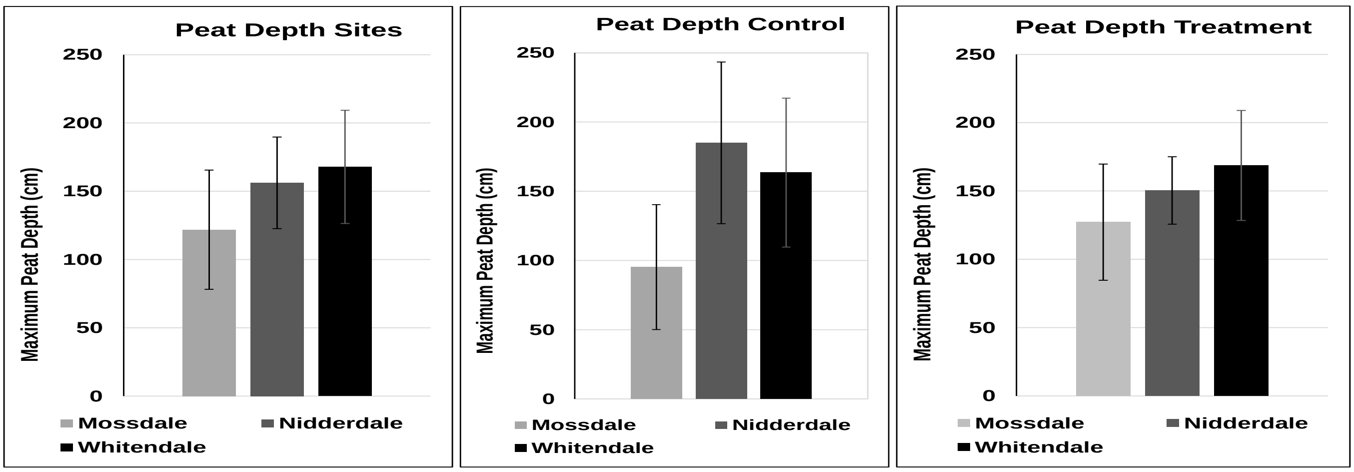


Fig. 12 Average maximum (i.e. to bedrock) peat depth (± standard deviation) based on manual pre-management measurements in July/August 2012 at the monitoring plots using the average of peat rod and peat core measurements per plot across all three sites (**left**), for the 'to be burnt' (Control) sub-catchments (**middle**) and for the 'to be mown' (Treatment) sub-catchments (**right**). For significant differences see text.

4.2.2 *Manual peat carbon stocks*

Peat organic carbon (C_{org}) stocks were assessed from manual peat cores taken with a 1.1 m box corer (see **Figure 13**) from each monitoring plot location in August 2012, before any management change. For all cores, 5 cm sections from a range of depths down the full peat profile were assessed for carbon content, using a C/N analyser, and for bulk density, to allow calculation of carbon densities. The sections sampled were (as far as the maximum peat depth allowed) 0-5 cm, 10-15 cm, 20-25 cm, 40-45 cm, 80-85 cm, 160-165 cm and 10 cm above the maximum peat depth. Values derived for these sections were assumed to be the same over the corresponding sections: 0-5 cm, 5-15 cm, 15-30 cm, 30-60 cm, 60-120 cm, 120-200 cm and 200 cm to the maximum peat depth (see Appendix 2 for further information on the methods).

Fig. 13 Example of a manual peat core taken using a 1.1 m box corer (**left**) with one side removed to allow access. Peat was extracted, cut into 5 cm sections and dried for detailed bulk density (**middle**) and carbon content analysis using a C/N analyser (**right**).

The detailed core analyses revealed bulk densities (**Figure 14**; left) that were well within the expected range for blanket peat (0.05 to 0.15 g cm^{-3}). At Nidderdale, there was an initial increase in the bulk density of the top 10-15 cm of peat, and whilst Whitendale showed a steady decline, Mossdale showed an increase up to 30-60 cm depth. All sites showed lowest bulk densities at about 60-120 cm depth followed by a sharp increase below 120 cm, which indicates increased compaction and a higher mineral content or possible C_{org} decline. While bulk densities showed clear site differences, specifically with lower values over the top 30 cm for Mossdale than Nidderdale and Whitendale, organic carbon content was fairly similar across sites (**Figure 14**; right), increasing from about 52% at the surface to about 58% towards the base, but declining sharply in the lowest sections. This decrease indicates the influence of higher levels of heavier mineral components, explaining the higher bulk density.

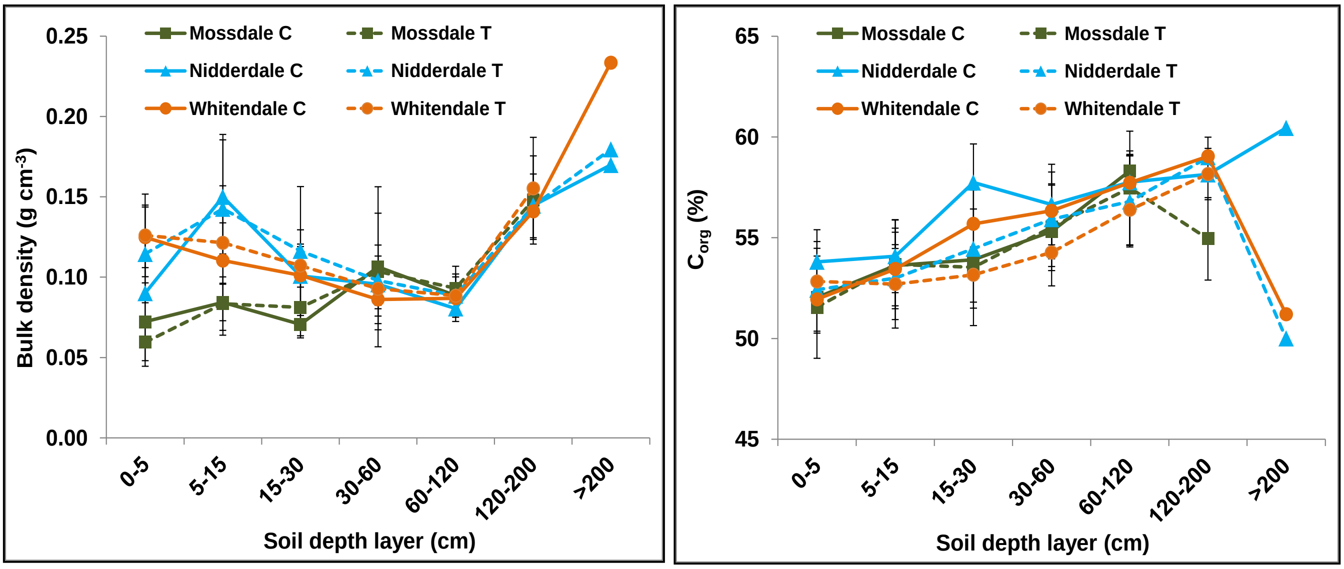


Fig. 14 Average (± standard deviation) bulk density (**left**) and organic carbon content (**right**) based on manual peat core section (i.e. 5 cm^3 sample of the mid soil depth range) measurements at the monitoring plots for each site split by sub-catchment (solid lines: burning control (C) sub-catchment; dashed lines: mowing treatment (T) sub-catchment) at the onset of the experiment before management change (30/08/12).

The carbon stocks, calculated using the bulk density and organic carbon content measurements from the peat cores described above, were significantly (ANOVA: $p < 0.001$, n = 24 per site; **Figure 15**) lower at Mossdale than at Nidderdale and Whitendale (mean ± standard deviation: 65.3±24.5, 80.8±19.4 and 95.4±25.2 kg C m^{-2}, respectively). Carbon stocks were also significantly ($p < 0.003$, n = 20 per site) higher in the Whitendale mowing sub-catchment compared to those of Nidderdale and Mossdale, but did not differ significantly between burning sub-catchments ($p > 0.05$, n = 4 per site), nor between the sub-catchments within sites ($p > 0.05$, n = 4 for burnt versus 20 for mown). These carbon stock estimates for this study were similar to the range reported by Garnett et al. (2001) for a blanket bog at Moor House (cf. 44.6 to 85.4 kg C m^{-2}).

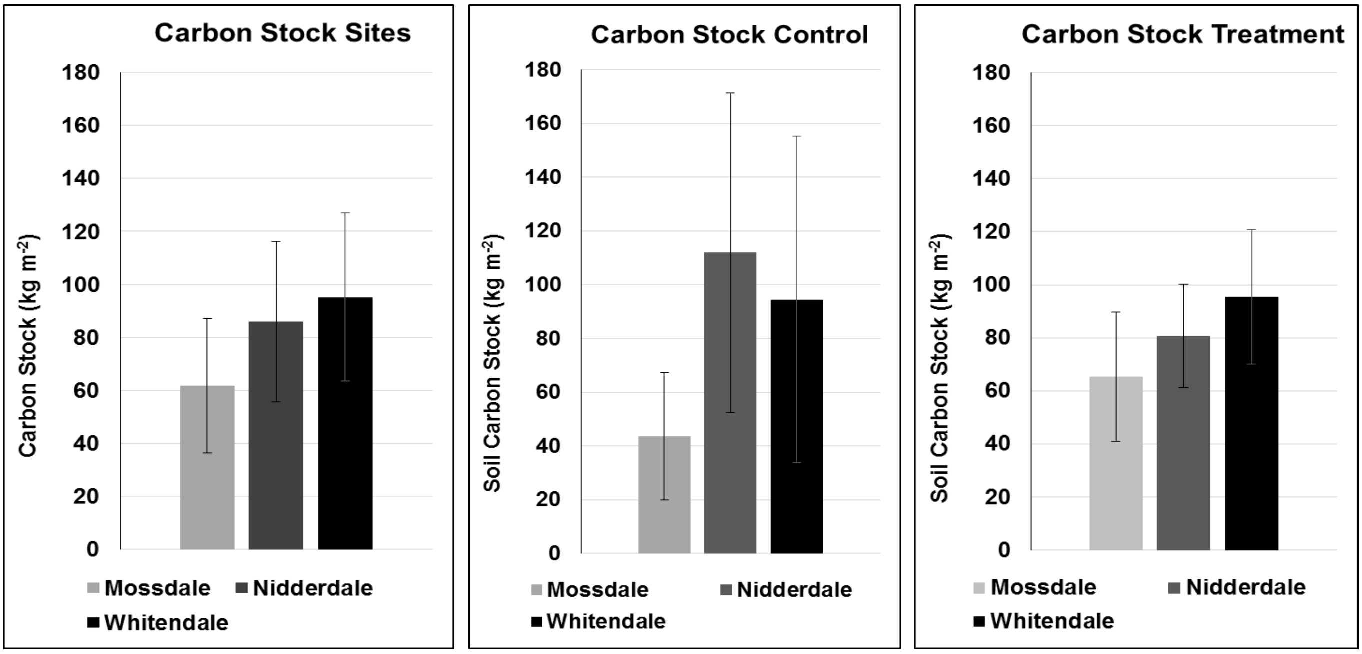


Fig. 15 Average (± standard deviation) of the soil (peat) carbon (C) stocks, calculated from the bulk density and organic C content of the full length of the peat cores extracted manually with the peat corer from the monitoring plots, for each of the three sites (**left**), the burning (Control) sub-catchments (**middle**) and the mowing (Treatment) sub-catchments (**right**) at the onset of the experiment before management change. For significant differences see text.

4.2.3 *Micro-topography and compaction assessment*

Fig. 16 Micro-topography monitoring (14-16/09/15) across the 5x5 m plots, by measuring the offset from the peat surface to a levelled line (same height above the peat surface at either end) across the plot (**left and centre**) and compaction assessment by taking 5 cm sections of surface peat at Whitendale on 26/03/13 for bulk density assessment (**right**).

The micro-topography (variation in peat surface) was assessed for each 5 x 5 m plot across all managements by measuring the height offsets (see **Figure 16** above) from the peat surface to a common level on 10 points for 10 transects per plot at all three sites. These measurements were made in September 2015, two years after management; see Appendix 9 for more detailed methodological information. An initial visual assessment had revealed initial compaction of the peat surface during mowing (i.e. the water table dipwells were pushed down by the tractors about 15-20 cm into the peat) which subsequently (on the same day) bounced back.

The offset variability (measured as the standard deviation of the measured offsets, which could be positive or negative) from the peat surface (**Figure 17**) was significantly (2-way ANOVA; $p < 0.001$) higher on burnt and uncut plots (mean of 5.8 cm), and noticeably lower for all the mown plots at all sites (mean of 3.8 cm), but without any observable impact by brash removal or *Sphagnum* addition (both of which required walking across the plots). However, at Whitendale, the burnt plots had a higher offset to the peat surface than the uncut plots, which were located in the much flatter adjacent mown sub-catchment (see elevation contour lines in **Figure 1**). Notably, the Whitendale burnt management area also had more gullies across the catchment (see aerial pictures in **Figure 2b**).

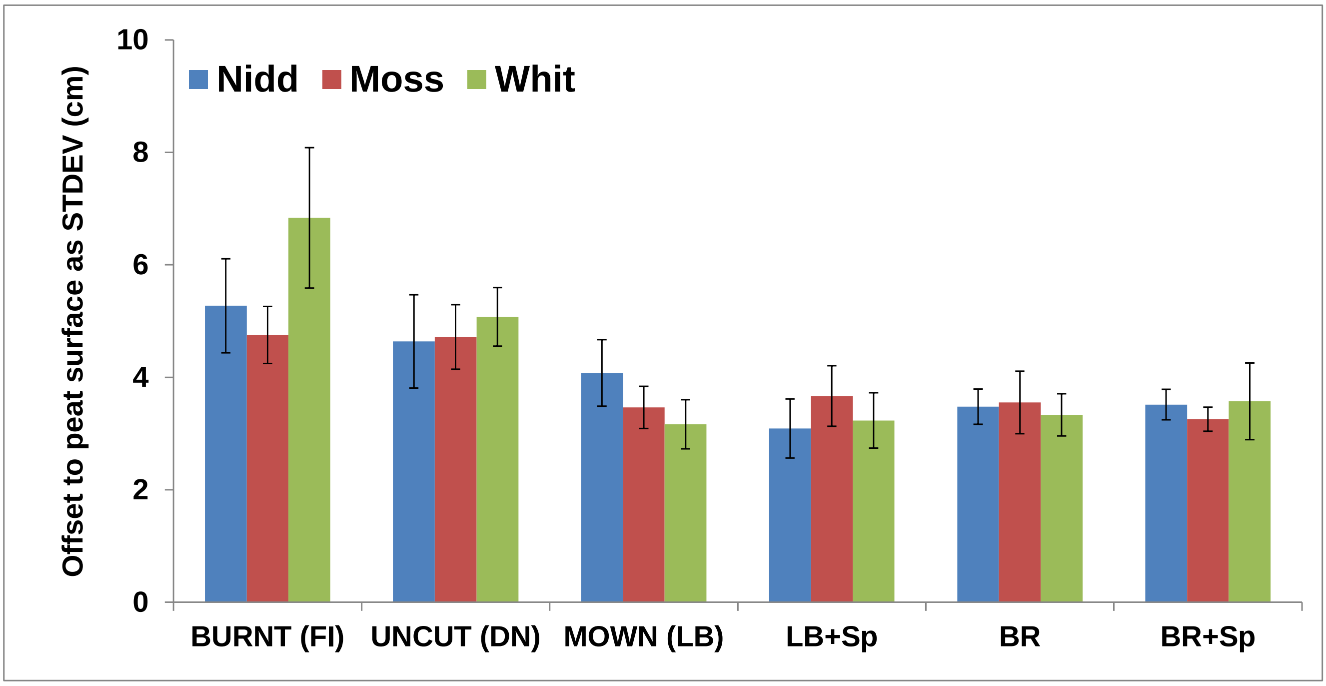


Fig. 17 Mean standard deviation (STDEV ± standard error; SE) of the measured offset (in cm) in relation to the average peat surface outside the plot area) from the micro-topography monitoring (14-16/09/15) across the 5x5 m plots for each management (n = 4 for burnt (FI), uncut (DN), mown with (LB) or without brash (BR) and with or without *Sphagnum* pellet (+Sp) addition) for Nidderdale (Nidd), Mossdale (Moss) and Whitendale (Whit).

Apart from the burnt plots at Whitendale, all three sites had median offsets of zero for uncut and burnt plots, whereas mown plots had more negative offsets, with lower interquartile ranges, than uncut and burnt plots (**Figure 18**). The before-after comparison is provided by comparing the managed plots to the uncut (in this instance the control) plots. Mowing impacts were slightly less at Nidderdale, where more basic and lighter mowing machinery (small tractor and a simple flail) was used, compared to the heavier machinery (heavy duty tractors with hammer flails and a fine double chop) at Mossdale and Whitendale. This provides strong evidence that plot micro-topography was made less variable, and the median offset reduced, through the mowing chopping off the tops of the hummocks.

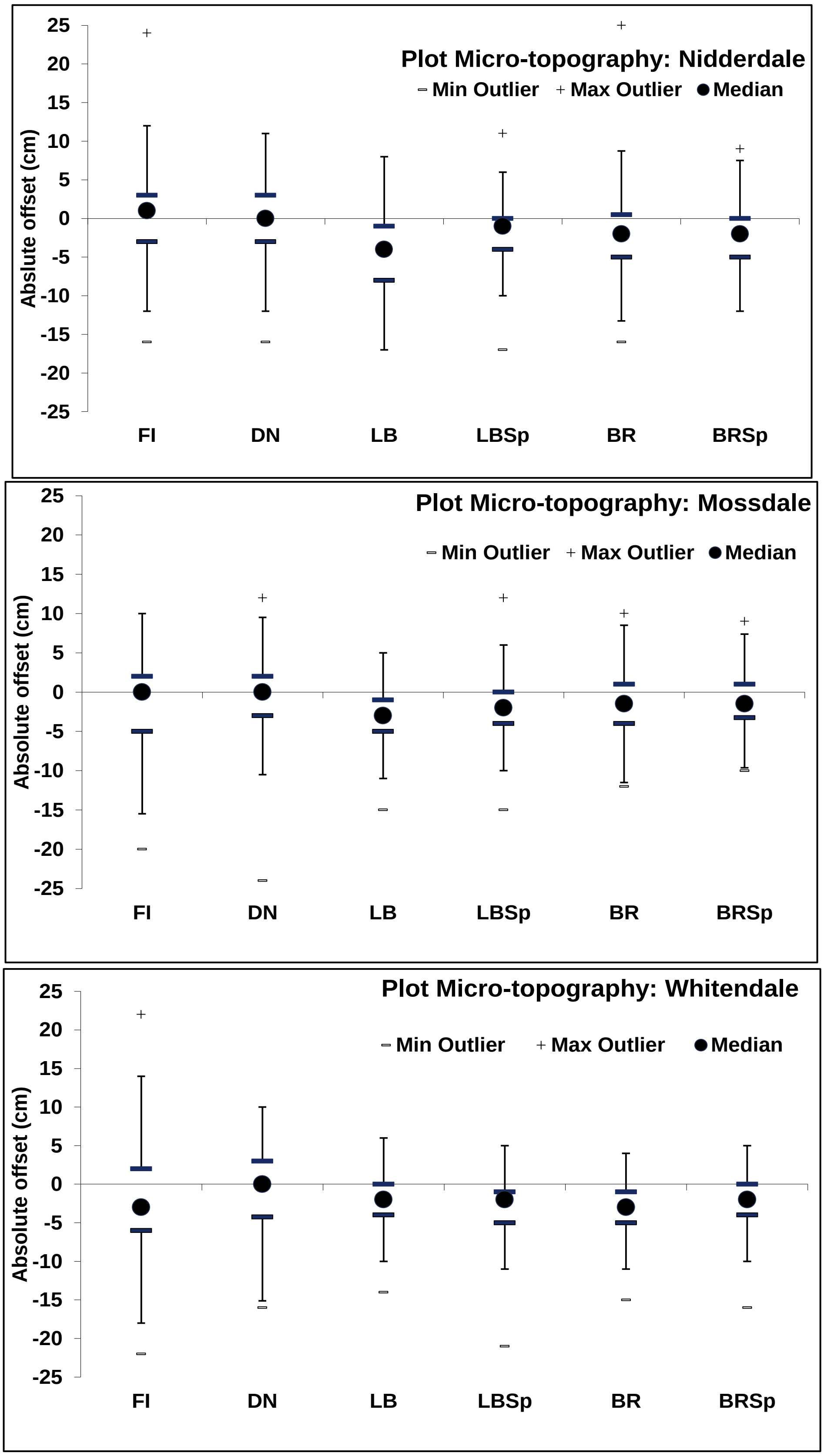


Fig. 18 Mean absolute offset (in cm compared to the average peat surface (zero) outside the plot area) from the micro-topography point monitoring (14-16/09/15) across the 5x5 m plots (45 points per plot) for Nidderdale (**top**), Mossdale (**middle**) and Whitendale (**bottom**) for each individual management (180 points per management: FI = burnt; DN = uncut; and combinations of LB = mown with brash left and BR = mown brash removed with or without Sp = *Sphagnum* addition). Negative values relate to dips, positive values to hummocks. Outliers are shown as + and – with median values as filled circles, the thick lines indicate the interquartile range and the thin lines are the upper and lower confidence range (1.5 times the interquartile range).

In April 2013, immediately after management, peat depth was measured near the water table depth loggers at all monitoring plots and compared to pre-management measurements taken in August 2012. Overall, peat depth (excluding hummocks) did not show any significant differences between pre- and post-management measurements at any site (**Figure 19**, neither for control (burnt; n = 4) or treatment (mown; n = 16) plots (i.e. excluding uncut plots). However, damage occurs less when the vehicle is moving straight than when standing still or tearing the surface such as when turning. Thus, damage is likely to be localised and hence may not have been picked up inside the monitoring plots.

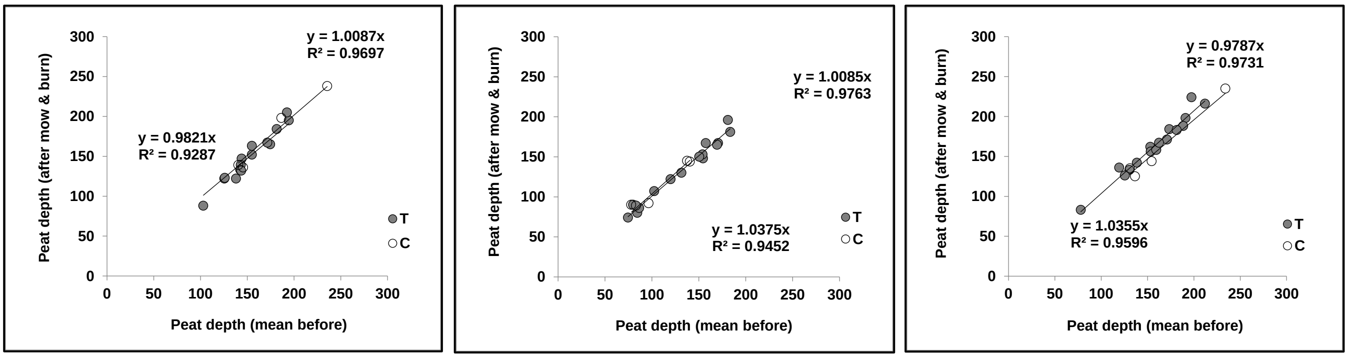


Fig. 19 Mean peat depth after management (29/04/13) (T = mown; C = burnt plots) plotted against that before management (29/08/12) for Nidderdale (**left**), Mossdale (**middle**) and Whitendale (**right**).

As effects on surface compaction might not be detected in a measurement of the full peat depth, an additional assessment of bulk density in surface peat layers (0 - 50 cm) was undertaken in March 2014 and compared to the initial assessment in August 2012 (see Appendix 2 for more detailed methodological information). A 5 x 5 cm three-sided box corer with a cutting blade (see Section 4.2.1; **Figure 11**) allowed for peat sampling without the cutting causing compaction of the peat and affecting bulk density. The 5 cm sections were taken *in situ* (see **Figure 16**) and bulk densities were assessed across four surface depths: 0-5, 10-15, 20-25 and 40-45 cm (see Section 4.2.2).

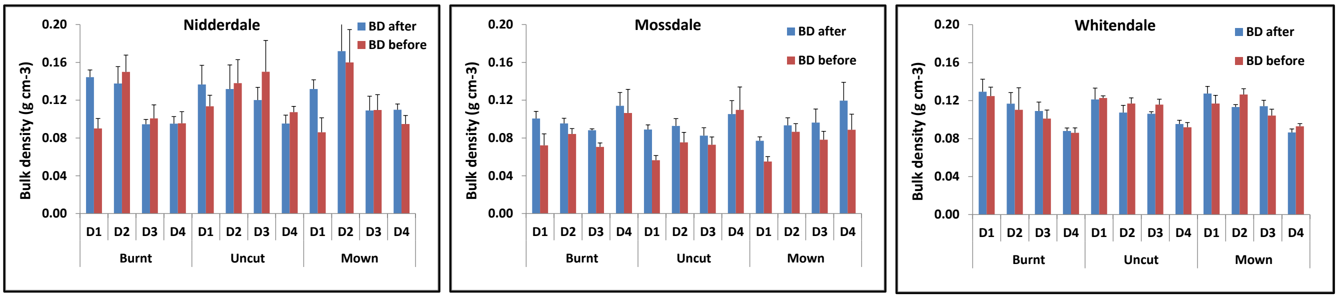


Fig. 20 Mean bulk density (+ standard error) measurements after management (25/03/14) (blue bars) and before management (29/08/12) (red bars) for burnt, uncut and mown plots at Nidderdale (**left**), Mossdale (**middle**) and Whitendale (**right**). Peat depths were: D1 (0-5 cm); D2 (10-15 cm); D3 (20-25 cm); D4 (40-45 cm).

The comparison of mean bulk densities (**Figure 20**) showed higher bulk densities after mowing management than before, particularly in the two upper peat layers (0-15 cm), at Nidderdale (mean±SE of 0.15±0.02 versus 0.12±0.02 g cm^{-3}, respectively) and Mossdale (mean±SE of 0.09±0.01 vs 0.07±0.01 g cm^{-3}, respectively), which might indicate compaction by mowing machinery. However, this was the case across all managements including the burnt and uncut plots, despite no tractors entering these sub-catchments or plot areas. Therefore, the observed change in bulk density was unrelated to management and reflected a natural process of bulk density change. A well-known natural process is “bog breathing” (Ingram, 1983) which causes peat level changes due to shrinkage and expansion in relation to water table and moisture changes (see Section 4.4.1). That this underlying natural cause explained the observed change in bulk density was supported by a larger apparent increase in bulk density at the surface for both Nidderdale and Mossdale (**Figure 20**), which could be related to rainfall and thus

peat moisture; whereas pre-management samples were taken after a particularly wet summer period in 2012, leading to expansion and thus lower peat bulk density, post-management samples were taken after a particularly dry spring in 2014 (see **Figure 10**), leading to peat shrinkage and thus higher bulk density. Whitendale peat bulk densities, unlike those at the other two sites, were remarkably similar between pre- and post-management assessments (mean±SE values for mown for the upper two peat layers were 0.12±0.01 g cm^{-3} for both periods) and importantly, this could be linked to similar moisture conditions in relation to high rainfall events (~60 mm) in mid-March 2014 (two weeks before post-management sampling), and a further ~30 mm of rain during the week of sampling, causing peat expansion.

In summary, despite the expected initial compaction of the peat surface after mowing, there was no lasting effect of management treatment on either peat depth or bulk density. However, plot micro-topography was made less variable by mowing chopping off the tops of some hummocks.

4.2.4 *Ground penetrating radar peat depth and peat pipe surveys*

Ground Penetrating Radar (GPR) surveys were carried out using automated, continuously monitoring, sledge-mounted equipment and around individual monitoring plots, using manually operated equipment (**Figure 21;** see Appendix 2 for more information). This provided data on variation in peat depth and carbon stocks across the catchments, as well as on peat pipe distribution. Whilst the GPR methods provided a comparison with the peat rod depth and carbon stock estimates (as described in Sections 4.2.1 & 4.2.2 above), peat pipes could only be detected by the manual GPR method (see Appendix 2 for further information).

Fig. 21 Example of the manual (**top**) and the automated (**bottom left;** Argocat pulling the automated equipment on a sledge) Ground Penetrating Radar (GPR) surveys used to detect peat pipes (**bottom right**) and peat depth at Whitendale.

The continuous GPR survey (August 2012) revealed peat depths (**Figure 22**) and allowed the estimation of soil carbon stocks (**Figure 23**) (using available bulk density and C_{org} data) across the sub-catchments, based on the interpolation (kriging) of the Argocat monitoring path (see **Figure 21**; left) across the sub-catchments containing the monitoring plot locations.

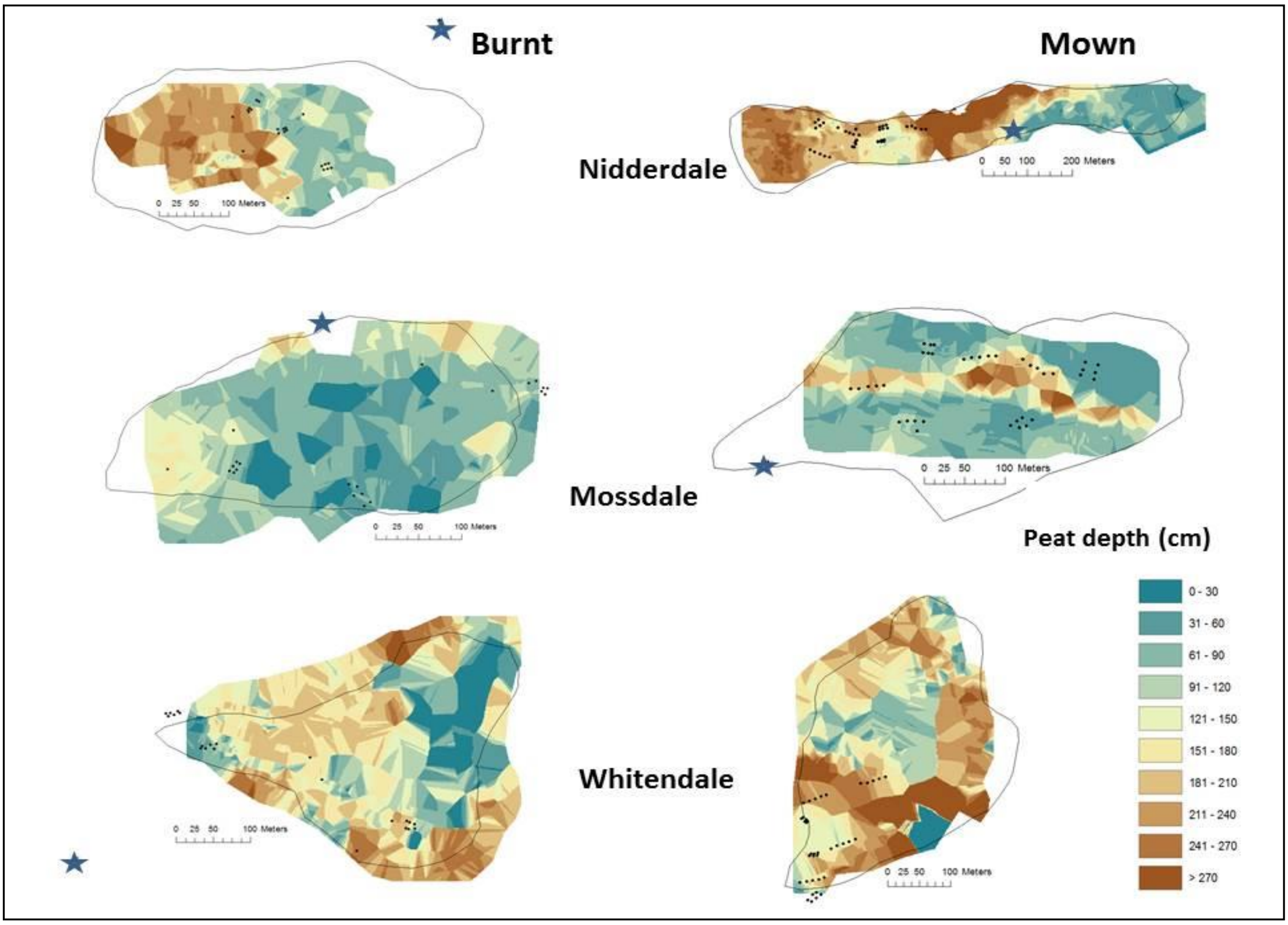


Fig. 22 Predicted (by kriging) peat depths across all six sub-catchments (black outline; **left** burnt; **right** mown) based on the interpolation of the automated GPR surveys. The weather station (star), plot (single or block of five) and slope (two rows of three) transect locations are also indicated as black dots.

The predictions (based on kriging) of the peat depth (**Figure 22** above) and soil carbon stocks (**Figure 23**) indicated that there was a fairly equal division of areas of higher and lower peat depth and carbon densities between each pair of sub-catchments (e.g. comparing Nidderdale mown versus burnt), with peat depths and carbon stocks generally being highest on flat areas and lower on sloping areas (for terrain see topographic indications in **Figure 1** and **Figure 2**). Overall, the Nidderdale and Whitendale sites showed much higher peat depths and carbon stocks across both sub-catchments than Mossdale.

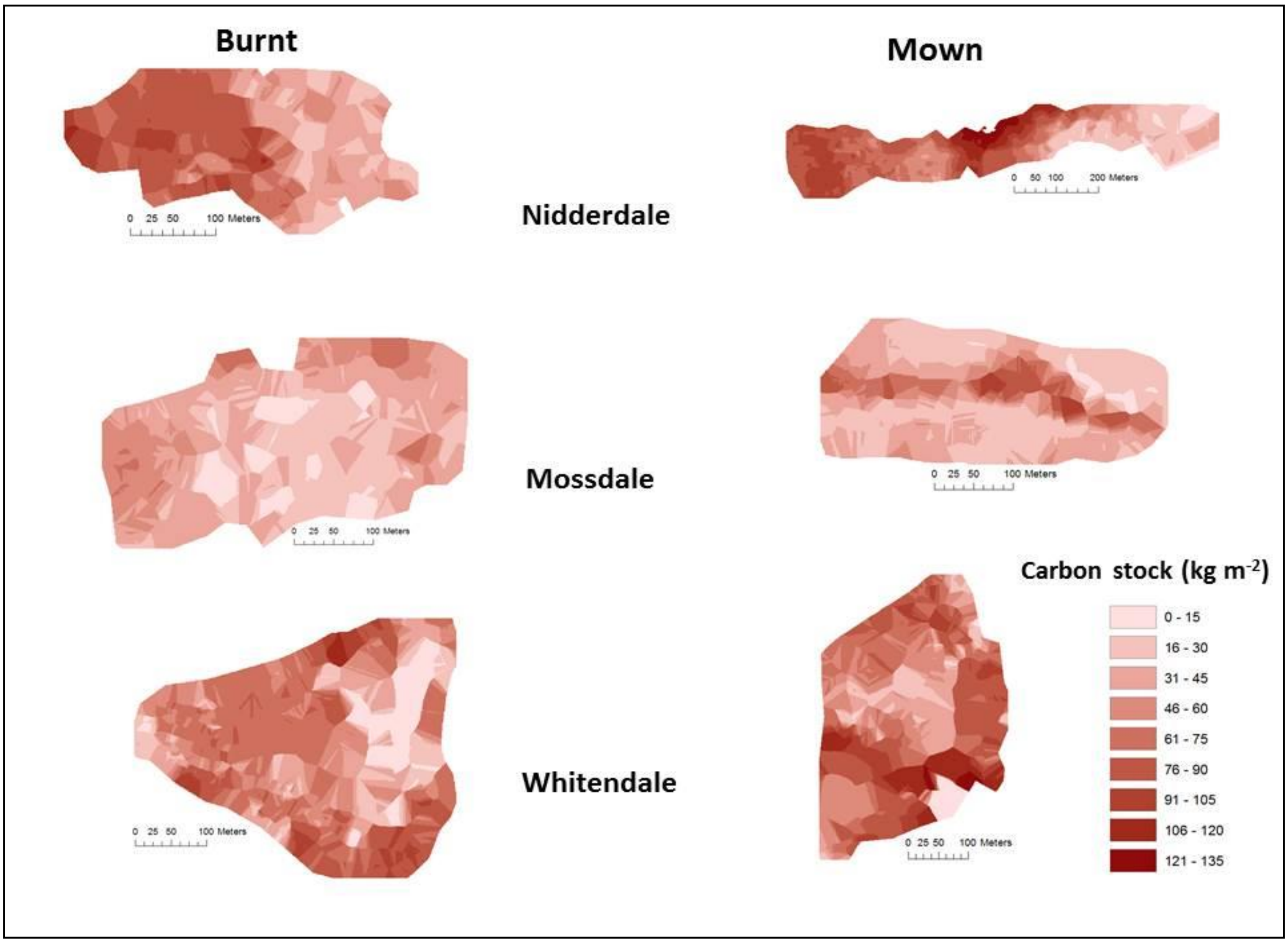


Fig. 23 Predicted (by kriging) carbon stocks (kg C m^{-2}) for the average maximum peat depth over the interpolated measurements for all six sub-catchments (**left** burnt; **right** mown) based on combined continuous GPR surveys for peat depth, manual coring surveys for bulk density and the percentage of soil organic carbon, which were all obtained before management change.

The area of the sub-catchments, the GPR measured peat depths, peat core bulk density and soil organic carbon content (averaged over the full core; cf. **Figure 14**) were used to calculate the soil carbon stocks per unit area (**Table 2**).

Table 2 Total peatland area and GIS-layer calculations (based on the automated Ground Penetrating Radar survey) of average (± standard deviation) peat depth, bulk density and the calculated area weighted mean soil carbon stocks for each sub-catchment of each site, where C is the to be burnt (Control) sub-catchment and T is the to be mown (Treatment) sub-catchment. Values were measured pre-management in August 2012.

Site/Catchment	Area (ha)	Peat depth (m)	Average bulk density (g cm^{-3})	Soil carbon stocks (kg C m^{-2})	Soil carbon stocks (kt C ha^{-1})
Nidderdale C	10.72	161.0 ± 70.3	0.077 ± 0.004	60.5 ± 24.0	0.60 ± 0.24
Nidderdale T	12.97	172.4 ± 96.6	0.076 ± 0.005	63.2 ± 32.3	0.63 ± 0.32
Mossdale C	8.64	81.9 ± 40.3	0.081 ± 0.002	32.6 ± 15.4	0.33 ± 0.15
Mossdale T	9.95	97.5 ± 55.9	0.080 ± 0.003	38.2 ± 19.8	0.38 ± 0.20
Whitendale C	8.21	140.8 ± 71.4	0.078 ± 0.004	53.4 ± 25.7	0.53 ± 0.26
Whitendale T	11.21	181.7 ± 79.5	0.076 ± 0.004	67.1 ± 26.3	0.67 ± 0.26

Up-scaled GPR-based soil C stocks (**Table 2**) reflected the differences in peat depth between sites (cf. **Figure 12**); estimated mean site soil C stock (across the full peat depth) was substantially lower at Mossdale (0.35 kt C ha^{-1}; 35.4 kg C m^{-2}) than at Nidderdale (0.62 kt C ha^{-1}; 61.9 kg C m^{-2}) or Whitendale (0.60 kt C ha^{-1}; 60.3 kg C m^{-2}). The mean values were similar between the paired sub-catchments, and (apart form Nidderdale) were substantially lower than peat core C stocks at the monitoring plots (**Figure 15**), reflecting larger catchment areas of shallow peat. The measured values remained comparable to the range of values measured for Moor House NNR (44.6 to 85.4 kg C m^{-2}; e.g. Garnett et al., 2001).

Peat depth values from the manual GPR survey (see **Figure 24** for an example output for each site) were comparable to those from the peat rod survey (**Table 3**); the mean depth of the peat rod survey usually fell within the minimum and maximum GPR-based values in 2012, clearly supporting the value of GPR surveys in capturing peat depth for experimental plots in much more detail and, importantly, being less destructive than any manual rod survey. Moreover, the GPR seemed to provide a depth resolution suitable for capturing depth changes and peat accumulation rates over time. Notably, the peat depth increase (i.e. difference between 2016 and 2012) for minimum and maximum GPR measured peat depths (**Table 3**) in our study (with similarly wet, high water table conditions of around -10 cm in 2012 and 2016 in the month before the surveys and thus limiting a 'bog breathing' effect) was on average 0.02 m over the five year period or about 0.4 cm yr^{-1}. This is similar to anticipated medium-term peat accumulation ranges (cf. 0.08-0.32 cm yr^{-1}) for heather dominated plots at Moor House over the last 50 years as reported by Hardie et al. (2007). The value is well above long-term accumulation rates at Moor House, which are about 0.04 cm yr^{-1} (Schillereff et al., 2016), and the long-term peat accumulation rates on these sites (0.12 cm yr^{-1}) determined in this project using peat layer dating (see Section 4.3.3). However, there was an indication (2-way ANOVA; $p = 0.066$) of a lower short-term peat accumulation of 0.08 cm yr^{-1} on burnt plots, likely reflecting no brash or litter accumulation, compared to accumulation on plots in the mown catchment of 0.58 cm yr^{-1}, which was significant for Mossdale ($p < 0.01$). Although this study indicates potentially valuable information to be obtained from GPR-based peat accumulation measurements, clearly longer timescales need to be considered.

Table 3 Summary of the minimum and maximum peat depth (m) obtained from the manual 5x5 m plot-level GPR surveys in 2012 (pre-management) and 2016 (post-management). Peat depths are averaged (±SE) for each site for the main managements of uncut (n = 4), burnt (n = 4), and mown (n = 8; combining the *Sphagnum* treatments) with (LB) or without brash (BR). Depths were compared to the single manual peat rod measurements (in bold) at each plot in 2012.

	Nidderdale					**Mossdale**					**Whitendale**				
	Peat depth (m) 2012			Peat depth (m) 2016		Peat depth (m) 2012			Peat depth (m) 2016		Peat depth (m) 2012			Peat depth (m) 2016	
Management	Min	**manual**	Max	Min	Max	Min	**manual**	Max	Min	Max	Min	**manual**	Max	Min	Max
Uncut	1.18	**1.32**	1.91	1.20	1.94	0.89	**1.26**	1.41	0.96	1.51	1.68	**1.77**	1.86	1.68	1.91
±SE	0.10	**0.09**	0.21	0.09	0.20	0.16	**0.25**	0.23	0.19	0.25	0.07	**0.07**	0.08	0.08	0.08
Burnt	1.45	**1.77**	1.79	1.45	1.84	0.69	**1.13**	1.06	0.70	1.03	1.23	**1.64**	1.79	1.23	1.78
±SE	0.27	**0.22**	0.21	0.27	0.19	0.13	**0.15**	0.19	0.12	0.18	0.26	**0.24**	0.28	0.25	0.25
Mown (LB)	1.48	**1.48**	2.02	1.54	2.06	0.97	**1.29**	1.38	0.99	1.41	1.53	**1.69**	1.80	1.55	1.80
±SE	0.09	**0.12**	0.12	0.09	0.12	0.15	**0.15**	0.18	0.16	0.17	0.11	**0.11**	0.10	0.11	0.11
Mown (BR)	1.33	**1.51**	1.80	1.31	1.85	1.06	**1.32**	1.40	1.10	1.43	1.47	**1.51**	1.73	1.49	1.73
±SE	0.09	**0.08**	0.09	0.10	0.09	0.14	**0.14**	0.15	0.14	0.15	0.19	**0.15**	0.17	0.19	0.17

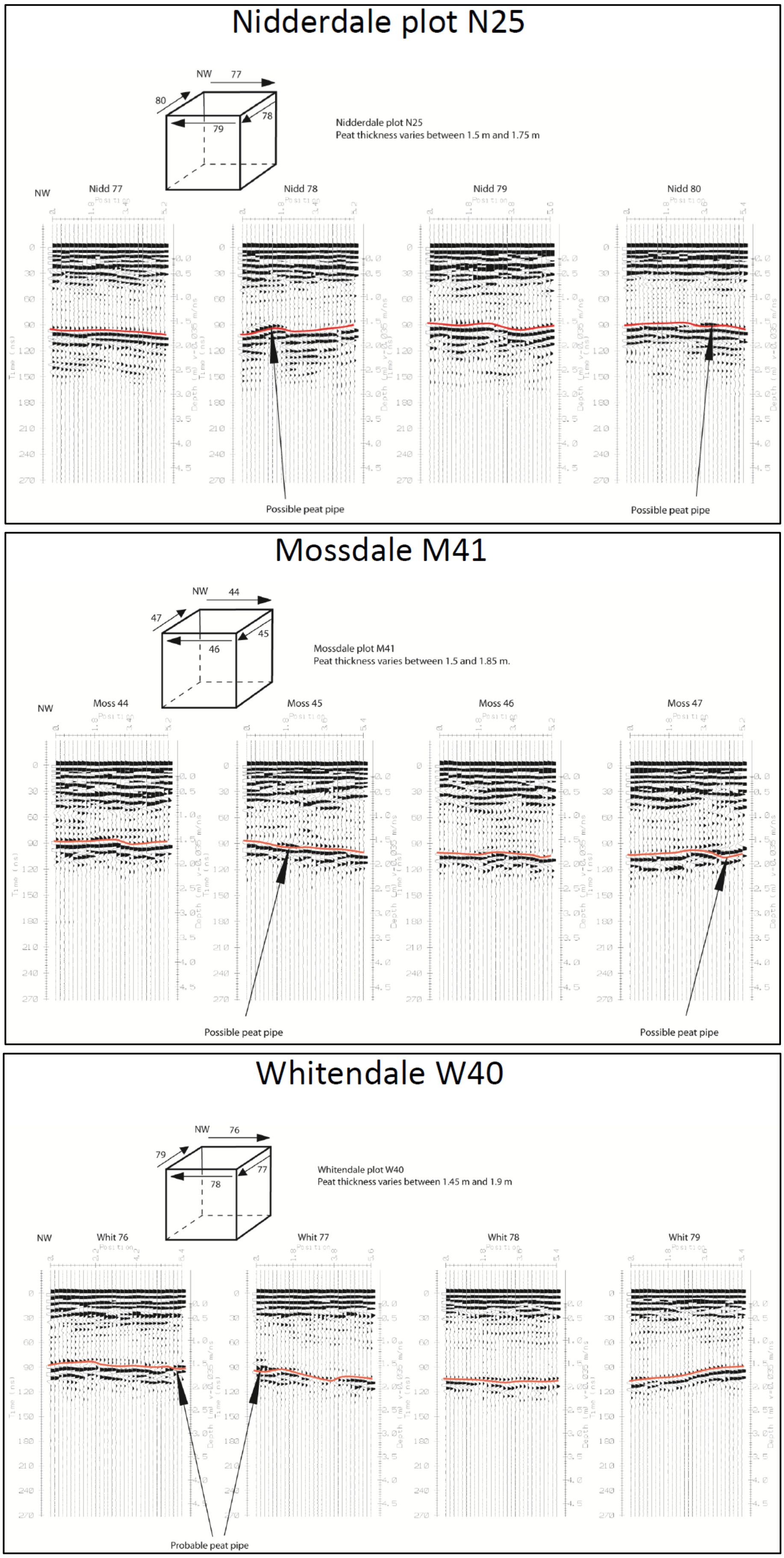


Fig. 24 Example of a GPR survey conducted during the pre-management period on 04/10/12 at Nidderdale (**top**), Mossdale (**middle**) and Whitendale (**bottom**) showing the peat depth (scale is in 30 cm sections down to a maximum depth of 270 cm) along all four sides of a plot (red lines). The arrows indicate likely peat pipes based on the separate scatter layer above the continuous scatter layer of the bedrock (below the red line).

The manual GPR survey (in end August 2012 and end March 2016) showed a much better signal resolution than that of the continuous survey, particularly enabling identification of possible peat pipes (**Figure 24**) and their occurrence across all monitoring plots at each site (**Table 4**). Peat pipe frequency was similar between sites and sub-catchments (**Table 4**). On average, there were a minimum of 0.53 identifiable peat pipes per 5 x 5 m plot in 2012 (an average of 12.7 across all 24 plots per site; see Appendix 2 for all plot data), with a minimum of 0.75 peat pipes per burning and 0.48 pipes per combined mowing sub-catchment plot. The average peat pipe occurrence per plot equates to 0.05 pipes per metre survey length (i.e. dividing the average pipe occurrence by 10 m, which is equal to two 5 metre sides of the surveyed plot square, assuming one entry and one exit point per peat pipe per square as shown in **Figure 24**), which compares well to an average of around 0.07 pipes per metre transect surveys found in a blanket bog GPR peat pipe survey by Holden (2005). Notably, most detected peat pipes were small (less than 10 cm in diameter), with only about 2-3 larger diameter pipes (of about 10-20 cm diameter) per site. Moreover, most pipes occurred at the peat-bedrock interface, indicating considerable potential drainage at the base of the peat. There was a slight increase (0.03 pipes per plot) in the number of peat pipes per plot between 2012 (pre-management) and 2016 (post-management). However, as the number of pipes did not change significantly over time, the small changes to date appear to be unrelated to the type of management.

Table 4 Summary of the minimum (Min; i.e. clearly identifiable), maximum (Max; i.e. including possible additional pipes) and number of large (Large; if so then included in Min) peat pipes obtained from the manual 5x5 m plot-level GPR surveys in 2012 (pre-management) and 2016 (post-management). Peat pipes are averaged (±SE if applicable) for each site (All) and main managements of uncut (n = 4), burnt (n = 4), and mown (n = 8; combining ±*Sphagnum*) with (LB) or without brash (BR).

	Nidderdale						**Mossdale**						**Whitendale**					
	Pipes (No) 2012			Pipes (No) 2016			Pipes (No) 2012			Pipes (No) 2016			Pipes (No) 2012			Pipes (No) 2016		
Management	**Min**	Max	Large	**Min**	Max	Large	**Min**	Max	Large	**Min**	Max	Large	**Min**	Max	Large	**Min**	Max	Large
All	**0.54**	1.00	2.00	**0.58**	0.00	1.67	**0.58**	1.00	1.00	**0.63**	1.00	1.00	**0.46**	1.00	1.00	**0.46**	1.00	1.00
±SE	**0.12**	-	-	**0.13**	-	0.12	**0.15**	-	-	**0.15**	-	-	**0.10**	-	-	**0.10**	-	-
Uncut	**0.75**	0.00	0.00	**0.75**	0.00	0.00	**0.00**	1.00	0.00	**0.00**	1.00	0.00	**0.75**	0.00	1.00	**0.75**	0.00	1.00
±SE	**0.25**	-	-	**0.25**	-	-	**-**	-	-	**-**	-	-	**0.25**	-	-	**0.25**	-	-
Burnt	**1.00**	0.00	2.00	**0.75**	0.00	2.00	**0.50**	1.00	0.00	**0.50**	1.00	0.00	**0.75**	0.00	1.00	**0.75**	0.00	1.00
±SE	**0.41**	-	-	**0.48**	-	-	**0.29**	-	-	**0.29**	-	-	**0.25**	-	-	**0.25**	-	-
Mown (LB)	**0.25**	0.00	0.00	**0.25**	0.00	0.00	**0.50**	1.00	1.00	**0.63**	1.00	1.00	**0.38**	1.00	0.00	**0.38**	1.00	0.00
±SE	**0.16**	-	-	**0.16**	-	-	**0.27**	-	-	**0.26**	-	-	**0.18**	-	-	**0.18**	-	-
Mown (BR)	**0.50**	1.00	0.00	**0.75**	0.00	1.50	**1.00**	1.00	0.00	**1.00**	1.00	1.00	**0.25**	0.00	0.00	**0.25**	0.00	0.00
±SE	**0.19**	-	-	**0.25**	-	0.25	**0.27**	-	-	**0.27**	-	-	**0.16**	-	-	**0.16**	-	-

4.2.5 *Heather assessment*

4.2.5.1 *Heather nutrition and carbon content*

Burning on grouse moor managed blanket bogs tends to encourage heather (*Calluna*) dominance, which is ultimately the aim, as heather is the main food source for red grouse. Burning is not only undertaken to increase the cover of heather but also to remove old growth, recycle nutrients and thus encourage new shoots of higher nutritious value (Lovat, 1911; Picozzi, 1968). Both nitrogen (N) and phosphorus (P) are important nutrients for red grouse (Moss, 1969; 1972), particularly for chicks (Savory, 1977) and breeding hens (Moss et al., 1975). There is, in fact, evidence that grouse selectively feed on heather with a high N and P content (Moss, 1972; 1977) and this selection is even more pronounced in chicks than adults (Savory, 1974; 1977). Moreover, grouse are also known to specifically require N, Na, Mg, Ca and K for egg laying (Moss, 1977), whilst a lack of Mn can cause breeding and development problems in poultry and pheasants (National Research Council, 1994). As grouse also prefer to consume heather shoots from plants between two and eight years old (Savory, 1978), management is necessary to encourage a constant supply of relatively young heather if the aim is to encourage grouse. However, relatively little is known about the effects of mowing on the nutrient value of heather for grouse, although there is evidence that heather regeneration after cutting is very similar to that after burning in terms of height and cover (Liepert et al., 1993). This study aimed to assess the effects of mowing on heather shoot nutrition compared to burning and uncut management.

Shoot samples were collected from all sites from net ecosystem exchange (NEE) CO_2 flux monitoring plots (**Figure 25**) before management in 2013, and again after management in 2015 (see **Table A3.1** for sample days and Appendix 3 for methods). Oven dried *Calluna* shoot subsamples (from the top 10 cm section) from the NEE plots were ground and then digested in 70% nitric acid. Filtered samples were run in an inductively coupled plasma mass spectrometer (ICP-MS; iCAP 7000 Series ICP spectrometer, Thermo Scientific, USA; **Figure 25**) to measure the following elements: P, K, Na, Ca, Mg, Fe, Al, Mn, Zn, Cu and Pb. Most Pb concentrations were below detection limit and Pb was therefore excluded from further analyses. Moreover, each ground oven dried *Calluna* sub-sample was also assessed for C/N concentration using the “Plant500” method in a C:N analyser (vario Macro, Elementar Analysensysteme, Germany). All elemental concentrations were calibrated using the appropriate standards (phosphorus or multi-element) and converted from ppm to $\mu g\ g^{-1}$ dry material or percentage of dry material using sample dry weight and any dilution factor. For more details on the analytical methods see Appendix 12.

Fig. 25 Assessment of heather shoots cut from a 30 cm diameter area (**left**) and scanned for the leaf area (**middle**) before elemental analysis for nutrient content by an Inductively Coupled Plasma (ICP) Mass Spectrometer (**right**).

Linear mixed models were implemented as described for plant species richness and diversity (Section 4.2.6; Appendix 3) with the following exceptions: the time periods, managements and sites were used as fixed effects in the mixed model and the only random effect used was the block the plots were located in as there was only one measurement post-management. More details on the statistical methods are given in Appendix 12.

The average C content of *Calluna* shoots was 52.3 (± 0.1 standard error) %. Although nutrient concentrations were broadly similar across sites (see **Table A12.1** in Appendix 12 for minimum, maximum and average values), concentrations of N, K, Na, Ca, Fe, Al, Mn, Zn and Cu in *Calluna* were significantly different between sites (see **Table 5** for P-values). However, there were no elements (apart from Na) for which the concentrations in *Calluna* differed significantly between managements within a site in the different time periods (see **Table 5**).

Table 5 Results from linear mixed effects models investigating differences in *Calluna* nutrient content between the four managements in the pre- and post-management periods (numerator df = 3) and between the four managements in each of the three sites in each period (numerator df = 6). For each model, the F value, the P value and the Satterthwaite approximation of the denominator degrees of freedom (df) are given. Significant values are shown in bold.

Element	Management: Period interaction		Management: Period: Site interaction		Site		Model df
	F value	P value	F value	P value	F value	P value	
N	12.28	**<0.001**	0.46	0.834	12.43	**<0.001**	119
P	5.09	**0.002**	0.94	0.469	2.70	0.071	115
K	8.06	**<0.001**	1.47	0.194	3.91	**0.023**	116
Na	3.76	**0.013**	2.51	**0.026**	12.25	**<0.001**	119
Ca	2.18	0.095	0.66	0.682	8.31	**<0.001**	116
Mg	3.00	**0.034**	1.06	0.393	1.89	0.156	115
Fe	1.44	0.234	1.63	0.147	9.85	**<0.001**	115
Al	3.69	**0.014**	2.05	0.064	8.88	**<0.001**	119
Mn	21.22	**<0.001**	0.78	0.585	3.60	**0.030**	119
Zn	2.68	**0.050**	2.06	0.064	18.59	**<0.001**	116
Cu	3.70	**0.012**	1.24	0.288	9.05	**<0.001**	299

Table 6 Results of post-hoc tests for pair-wise comparisons of elemental concentrations between the managements are given, where DN is uncut, BR is brash removed, LB is left brash and FI is burnt. Only post-hoc results from the post-management period are shown. Significant values are shown in bold.

Element	DN to BR	DN to LB	DN to FI	BR to LB	BR to FI	LB to FI
N	**<0.001**	**<0.001**	**<0.001**	0.995	0.403	0.117
P	**<0.001**	**<0.001**	**<0.001**	0.844	0.262	0.930
K	**0.036**	**0.009**	**<0.001**	0.999	**<0.001**	**<0.001**
Na	0.143	0.074	0.099	0.999	0.999	1.000
Mg	**0.005**	**<0.001**	**<0.001**	0.899	0.092	0.646
Al	0.062	**0.009**	**0.031**	0.995	0.996	1.000
Mn	**<0.001**	**<0.001**	**<0.001**	1.000	0.554	0.460
Cu	0.999	0.998	0.855	1.000	0.950	0.978

Across all sites nutrient concentrations in *Calluna* shoots were more similar between managements before management implementation than after (see **Figure 26a,b;** and see **Table A12.2** in Appendix 12 for minimum maximum and average values). Concentrations of N, P, K, Na, Mg, Al, Mn and Cu showed a significant interaction between management and time periods (see **Table 5** for P-values). Of these eight elements, there were no significant differences between management for Na, Zn and Cu in the post-management period (p> 0.05 for all); results of the post-hoc tests for the post-management period for the elements with a significant management

and time period interaction are summarised in **Table 6**. Uncut (DN) plots had significantly lower concentrations of N, P, and Mn ($p< 0.001$), Mg ($p< 0.005$), and K ($p< 0.05$), than all other managements (**Table 6**; **Figure 26a,b**) and significantly higher concentrations of Al than mown with left brash (LB; $p = 0.009$) and burnt (FI; $p = 0.031$) plots. There was no significant difference in N, P, Mg, Al and Mn concentrations between brash removal (BR), LB and FI plots, but burnt plots had significantly ($p< 0.001$) higher K concentrations than either mown BR or LB plots (**Table 6**; **Figure 26a**). The only element which showed a significant ($p = 0.026$) interaction between management, time period and site was Na (**Table 5**; **Figure 26a**) but the post-hoc tests revealed no post-management differences (**Table 6**).

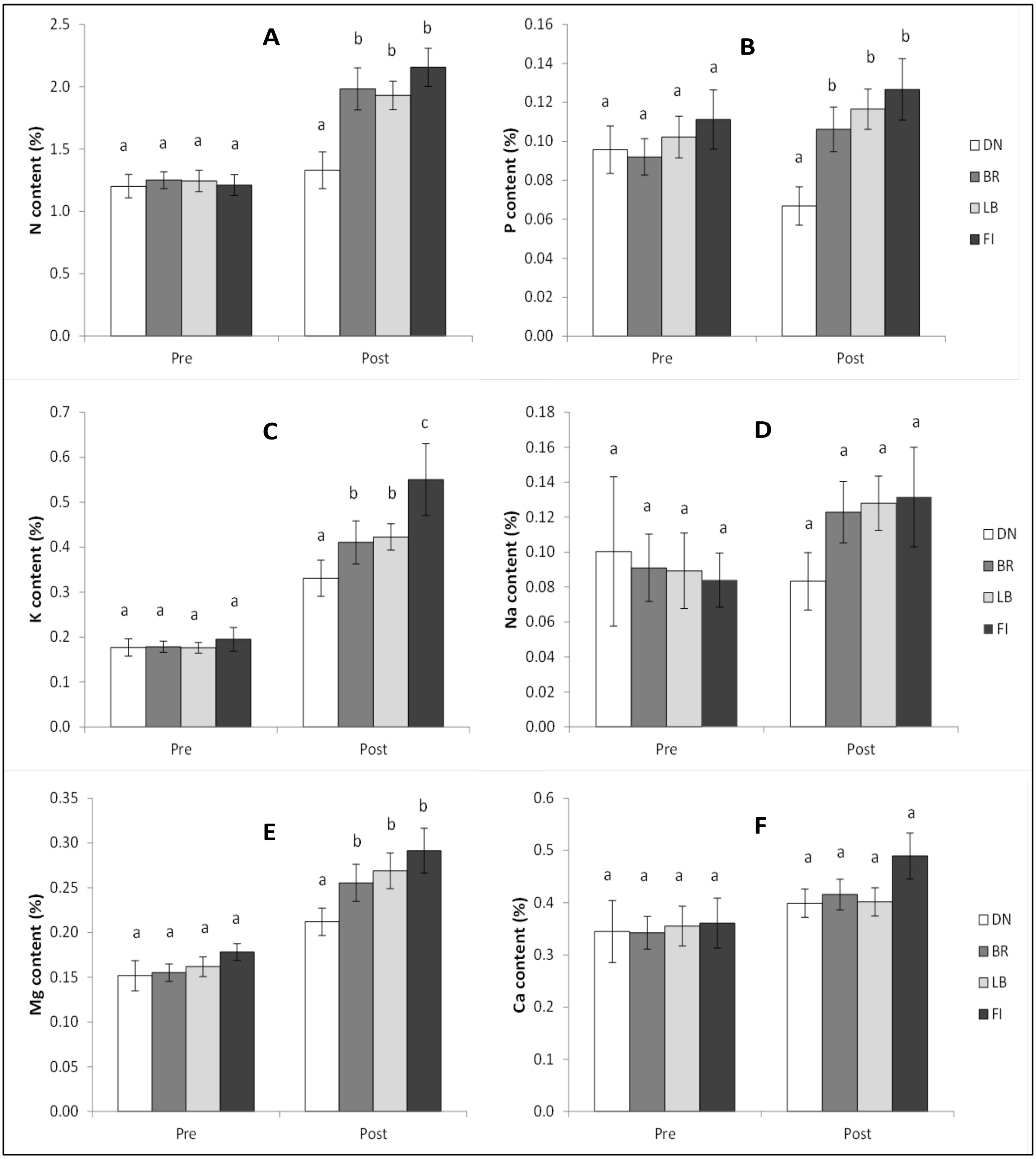


Fig. 26a Mean (± 95% confidence interval) percentage values of **A**) N content, **B**) P content, **C**) K content, **D**) Na content, **E**) Mg content, **F**) Ca content of *Calluna* shoots for each management group in the pre- (2012) and post-management (2015) periods. Management codes were DN (uncut), BR (brash removed), LB (left brash) and FI (burnt). Different lower case letters within each time period indicate significant differences between managements.

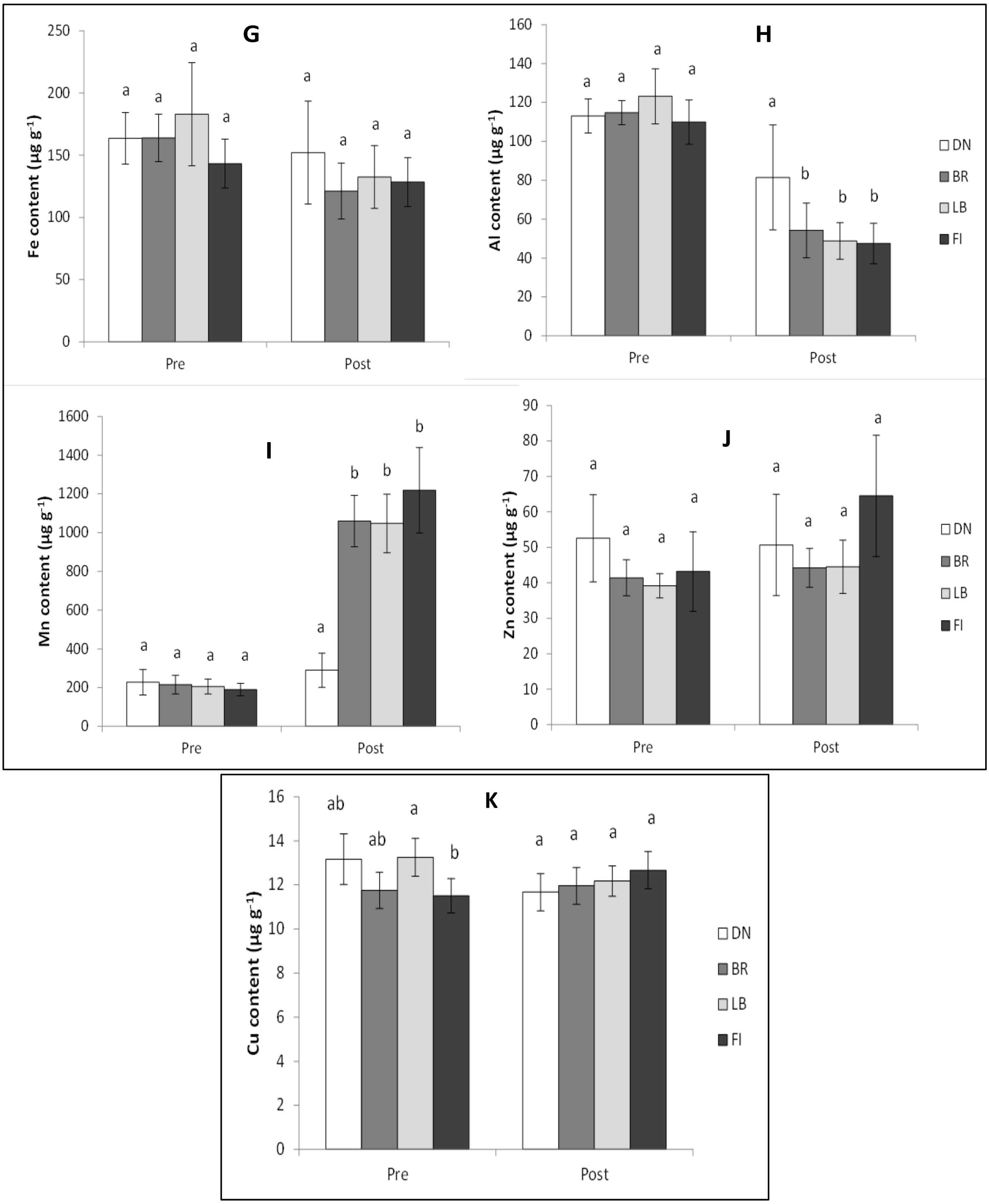


Fig. 26b Mean (± 95% confidence interval) values of **G**) Fe content, **H**) Al content, **I**) Mn content, **J**) Zn content and **K**) Cu content of *Calluna* shoots for each management group in the pre- (2012) and post-management (2015) periods. Codes were DN (uncut), BR (brash removed), LB (left brash) and FI (burnt). Different letters within each period indicate significant differences between managements.

Discussion

Moss (1967) noted that there was no change in chemical composition of *Calluna* over 6 years old although it did vary with the season. As seasonal variability is probably due to climatic conditions in the locality, this could at least partially explain some of the differences in the *Calluna* nutrient content between sites. This may also explain the small differences in *Calluna* nutrient content between pre- and post-management DN plots. The most striking revelation in this part of the project was that, of the eight elements which differed significantly between management, seven (N, P, Na, Mg, Mn, Al and Zn) did not show any significant difference between the burnt (FI) and either of the mown (BR and LB) management post-management (**Table 5** and **6**). Additionally, N, P, K, Mg and Mn had significantly lower content in *Calluna* from uncut (DN) plots than from the managed plots. The significantly lower P content of *Calluna* on uncut plots in the post-management period (**Figure 26a** B) clearly indicates that *Calluna* age affected P uptake (Moss, 1969). Therefore, it may not be advisable to leave heather completely unmanaged if grouse production is a concern, although natural layering of old *Calluna* under moist and mossy conditions (e.g. Mossdale) could rejuvenate leading to a mean stem population age of around 12-15 years as observed at Moor House NNR (Rawes & Hobbs, 1979) without the need of any management. Additionally, Savory (1978) showed that grouse preferred to eat shoots from between two and eight year old heather and, from the results shown here, either management is more likely to provide this than the uncut treatment.

The only element for which the *Calluna* differed significantly between burnt and mown plots was K (**Figure 26a** C). Estimates of the amount of *Calluna* eaten by red grouse range from 63 g d^{-1} for wild cocks to 100 g d^{-1} for wild laying hens (Savory, 1974). Laying hens require higher amounts of some elements due to the nutrients being used in the creation of eggs (Jenkins et al., 1965). A wild grouse during egg laying retains (i.e. requires) up to 160 mg K d^{-1} (Moss, 1977). Therefore, even though *Calluna* from the burnt plots had significantly more K than the brash removal (BR) mown management, a grouse eating from the BR management plots would only need to eat up to 39 g of *Calluna* each day to satisfy its K requirements. Even eating *Calluna* from uncut (DN) plots, which had significantly lower concentrations than all the other managements, the grouse would only require 48 g of *Calluna* per day. This shows that even grouse eating only 63 g *Calluna* d^{-1} would easily consume the necessary amount of K, regardless of the management.

Likewise, using the maximum values required by a laying hen (which are assumed to be the maximum amounts required and may also be instrumental in breeding success; Jenkins et al., 1965) reported by Moss (1977) for each element, *Calluna* from any management would satisfy the hen's N, Na, Mg and Ca requirements. Uncut (DN) plots had lower concentrations of all these elements but even so, a laying hen which solely fed on unmanaged areas would at most require 50 g of *Calluna* to satisfy its N requirements, 31 g for Na, 11 g for Mg and 80 g for Ca. For P, a hen would require 129 g of *Calluna* if solely feeding on uncut plots, which is substantially greater than the upper estimate of dry matter consumed per day (Savory, 1974). However, 68 g, 81 g and 74 g of *Calluna* would be require to meet P requirements if feeding was solely on burnt, mown with or without brash removal plots, respectively, all of which are within the range of *Calluna* consumed per day. This suggests that some form of management is required to recycle some of the P from older plants into new growth, making it available to red grouse, although in the long-term regrowth via natural layering on unmanaged areas might also provide adequate conditions for grouse.

There is evidence that red grouse selectively consume plants with higher N and P contents (Moss, 1972; 1977) and this selection is even more pronounced in chicks than adults (Savory, 1977). Chicks selected *Calluna* with an average of 2.32% N and 0.25% P (Savory, 1977). These values are much higher than the average values on any management in this study, although seven plots (out of the 72) contained more N than this. However, chicks are likely to only eat the very newest tips (Savory, 1977) and the material analysed for burnt and mown plots contained a mixture of leaves representing three years of growth. Also, the post-management *Calluna* was cut in

August whereas pre-management *Calluna* was cut in early spring, which likely affected the content of some nutrients (Moss, 1967), although it should not have affected the differences between managements within a period. Additionally, although *Calluna* is the main component of the red grouse's diet, both chicks and adults eat other plants and some insects (Savory, 1977). Where available, *Vaccinium myrtillus* is sought after, which has a higher P content and usually begins growing earlier in spring than *Calluna* (Moss, 1972). Few studies on red grouse nutrition have been conducted on moors with high *Eriophorum* cover, or if they have, the nutrient content of *Eriophorum* has not been considered although flower buds of *Eriophorum* spp. have been shown to be a high N and P food (Pulliainen & Tunkkari, 1991). Therefore, it might be worth land managers considering the range and cover of other species alongside *Calluna* when considering management options to benefit their grouse populations. The positive management effect of either burning or mowing in a 3-fold increase in Mn content is of particular importance. A lack of Mn can cause breeding and development problems in poultry and pheasants (National Research Council, 1994), although as with N and P, it appears that it is management in general which is required to elevate Mn concentrations rather than specifically mowing or burning.

4.2.5.2 *Heather volume and biomass component assessment*

Heather plants can reach substantial size with a considerable shoot biomass, possibly taking up a considerable part of the gas flux chamber volume. As gas flux (e.g. CO_2) calculations are based on the chamber volume, it could be important to correct for plant sizes affecting the actual internal chamber gas volume. Therefore, plant volumes were estimated in the field and these estimates were validated in the laboratory. In 2012, the volume of heather within each net ecosystem exchange (NEE) flux measurement circle (see **Figure 27**) was assigned a volume category between 1 and 10 (corresponding to ~10% and 100%, respectively) when the NEE chamber (area of 700 cm^2, height of 60 cm and a volume of 39.6 L) was in place. Shortly before burning and mowing in spring 2013, all plants were cut at the stem base, bagged and sealed to allow assessment of plant volume, measured by water displacement in a bucket. Similarly, regrown heather shoots (i.e. on burnt or mown plots) were cut and measured in 2015. These were also arranged on foam mats (**Figure 27**), which allowed assessment of the percentage of the chamber area the heather covered, which was needed for upscaling the measured heather biomass to the entire 1 x 1 m and 5 x 5 m plot scales. Appendix 3 provides full details of the methods.

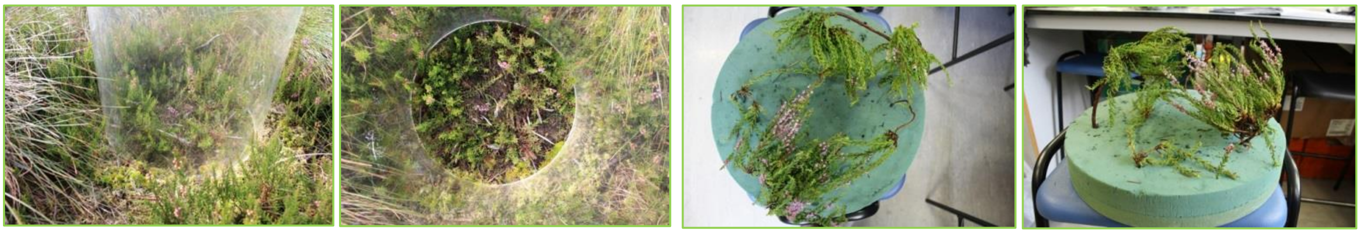


Fig. 27 Heather (*Calluna*) volume assessment in 2015 for individual net ecosystem exchange plot areas in the field (**two on the left**) and for samples cut from equivalent areas within the plots (**two on the right**). Samples were placed on a foam mat to allow visual scoring of height and covered area (using digital pictures) and volume categories, similar to those made in field assessments. Volumes were also determined by water displacement of the samples.

The aims were to verify whether including plant volumes in NEE flux calculations was necessary and, if so, whether volume estimates made in the field could be used as proxy for volume measurements, without the need for destructive sampling. Heather volume taken from the 660 cm^2 of NEE plot area (i.e. 29 cm diameter circles) varied mostly between 500 to 2500 ml with differences between sites in shoot and stem weight (**Figure 28**).

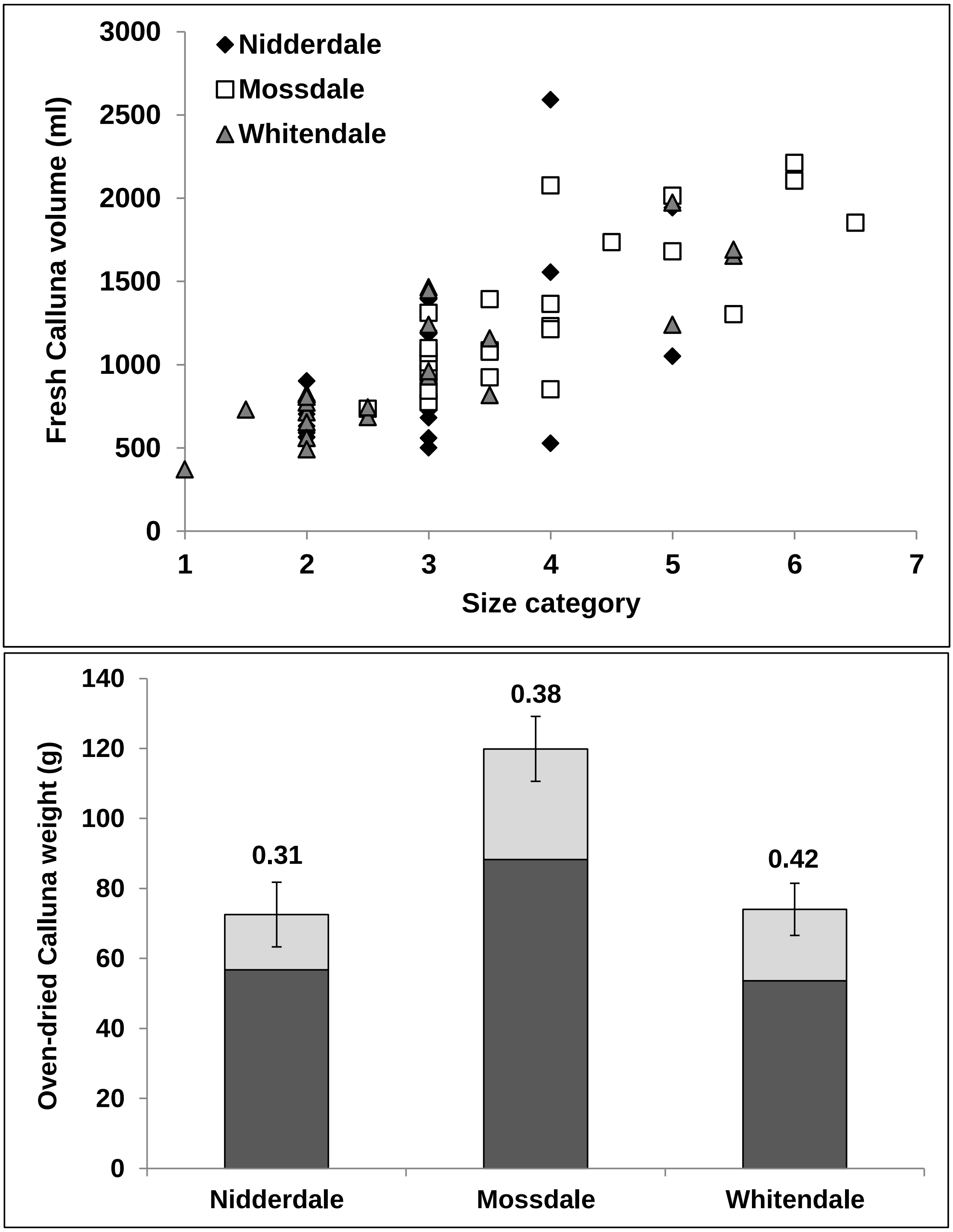


Fig. 28 *Calluna* volume versus size category (**top**) and mean (± standard error; SE) oven-dried *Calluna* dry weight (**bottom**) for green shoots (light grey) and woody stems (dark grey) for *Calluna* shoots harvested from net ecosystem exchange (NEE) plots in spring 2013 (shortly before management). Average shoot to stem ratios are given for each site above the SE bars.

The estimated percentage of chamber volume occupied by *Calluna* (but measured in categories) ranged from 10% to 65% with a mean of 33%. The percentage of the actual chamber volume occupied by *Calluna* ranged from 0.93% to 6.54% with a mean of 2.73%, which was approximately 12 times less than the estimated percentage volume. Mossdale had the greatest plant volume on average (1.33 L) and Nidderdale had the smallest (0.94 L). Importantly, there was a significant linear relationship between the estimated and measured volumes across the sites (R^2 = 0.60, p< 0.0001) and at the three sites individually, which was weakest at Nidderdale (R^2 = 0.33, p< 0.0001), stronger at Mossdale (R^2 = 0.63, p< 0.0001) and strongest at Whitendale (R^2 = 0.73, p< 0.0001), likely reflecting 'training' of the observer, as volumes were determined first at Nidderdale and last at Whitendale.

On average, the dry biomass of the *Calluna* cut from the chambers pre-management (2013) was 88.8 ± 11.0 g and did not differ significantly between managements (**Figure 29**; χ^2_3 = 0.89, p = 0.83). Post-management, the average *Calluna* biomass cut from the 30 cm diameter chamber-sized circles on DN plots was 97.0 ± 24.9 g which was significantly higher than the 6.0 ± 1.4 g cut from burnt and mown plots (**Figure 29**; χ^2_3 = 29.65, p< 0.001). When post-management *Calluna* dry biomass (in 2015) was scaled up (considering heather cover) to the full plot sizes, there was 9,737 ± 2,890 g on the 5 x 5 m DN plots and an average of 639 ± 130 g on the BR, LB and FI plots (these managements were combined as values were very similar). On average, the pre-management FI plots contained 1,129 ± 297 g *Calluna* biomass m^{-2} with a C content of the corresponding 'leafy' (i.e. stems with green leaves) material of 52.3 ± 0.1 % as determined by elemental analysis (see Section 4.2.5.1) and, assumed to be similar or greater in 'woody' (i.e. stems without green leaves) material, this meant that at least about 587 ± 155 g C m^{-2} was lost on average during burning.

The average 'leafy' to 'woody' ratio (i.e. in grams of material) was over five times higher post-management than pre-management (**Figure 29**). The 'leafy' to 'woody' ratio was only significantly different between managements after management implementation (χ^2_3 = 30.10, p< 0.001), with DN plots having substantially lower ratios than either burn or mowing management (p< 0.001 for all). Leaf area index (LAI) was higher overall in the pre-management period and showed no differences between managements in this period (Figure 35; χ^2_3 = 0.76, p = 0.86). Similarly to the 'leafy' to 'woody' ratio, LAI (**Figure 29**) was significantly different post-management (χ^2_3 = 28.74, p< 0.001) with uncut plots having higher LAIs than any of the managements (p< 0.001 for all). *Calluna* on uncut (DN) plots had a far lower 'leafy' to 'woody' ratio than the mown (BR and LB) or burnt (FI) plots post-management (**Figure 29**), reflecting older and therefore less vigorous growth (Gimingham, 1975) than the younger plants, as well as possessing longer and thicker lignified stems. A higher 'leafy' to 'woody' ratio may seem beneficial for grouse but the quantity of *Calluna* leaves was more than seven times greater on uncut plots than any other management. However, the amount of new, and therefore more nutritious, shoots (see Section 4.2.5.1) was very similar for mown and burnt plots. Mowing, therefore, offers a viable alternative to burning on the basis of providing nutritious food for grouse. Another benefit of management is that most of the unproductive woody growth was removed, lowering the potential fuel load and potentially reducing the risk of wildfire impacts (Allen et al., 2013).

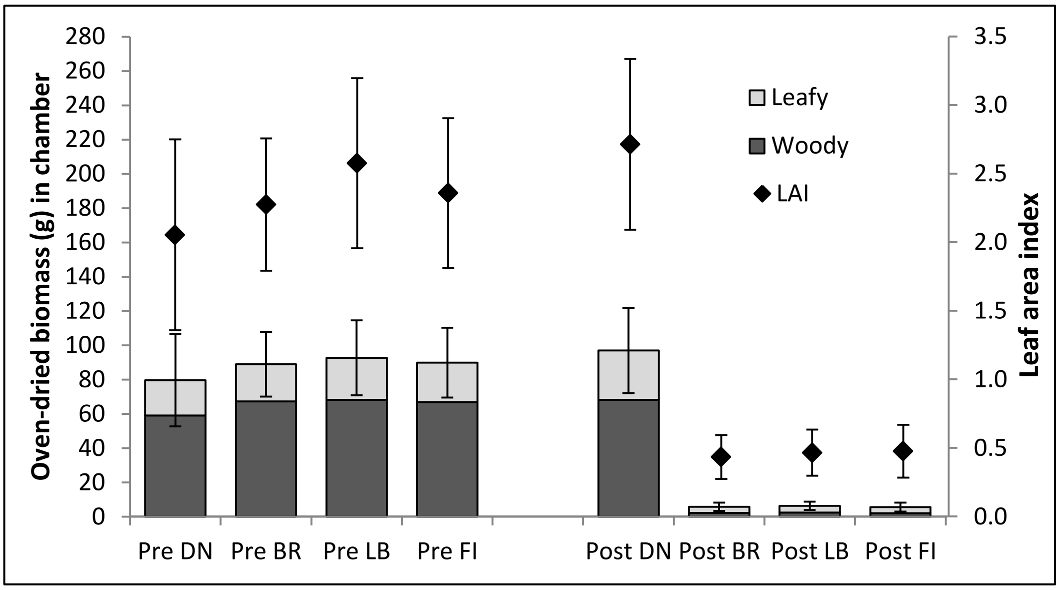


Fig. 29 Mean (± 95% confidence intervals of the mean) of the biomass (split into 'leafy' and 'woody' biomass, represented by pale and dark grey bars respectively) and leaf area index (LAI; black diamonds) of the *Calluna* biomass cut from 30 cm diameter areas (covering an area of 660 cm^2) for each management group, from the actual net ecosystem exchange (NEE) in the pre-management (2013) and equal 30 cm diameter areas for the post-management (2015) period (see Appendix 3 for dates).

Additionally, management significantly reduced the *Calluna* LAI (**Figure 29**) which therefore opened up the canopy to the benefit of other vascular plants and mosses. Typically, LAI values at Moor House NNR, a much-studied bog with small experimental areas managed by burning, were between 1 and 2 (Heal & Perkins, 1978), which were low compared to the post-management uncut (DN) plot values (2.71). However, Moor House plants might have been older, thus opening up gaps by falling over, as reported for *Calluna* older than 30 years (Gimingham, 1975). The age of *Calluna* in the current study was unknown, but was assumed to be similar across sites with an estimated age of 15-20 years in 2012 based on its height and shoot branching. Although overall burnt and mown *Calluna* did not differ post-management (**Figure 29**), burnt plots initially had shorter plants than mown plots at both survey scales (see **Figure 33** in the following Section 4.2.6) despite not all differences being statistically significant. This clearly related to the slower growth rates of plants which germinated from seed (largely on burnt areas) as opposed to those re-growing from the original stem (largely on mown areas). Liepert et al. (1993) also found that burnt areas had more *Calluna* seed germination whereas mown areas had more vegetative regrowth. However, long-term impacts of regrowth are unknown and could be very different. Whereas burning provides easily accessible ash fertilisation, mowing only provides nutrients from slower brash decomposition and potentially provides less overall as, over time, organic matter is incorporated into the peat. Only continuation of monitoring of the regrowth over a complete rotation cycle would allow assessing such a potential management difference in fertilisation effect and overall regrowth.

4.2.6 *Vegetation composition*

Overview of methods

Vegetation was assessed annually in early to mid-autumn (see **Table A3.1** in Appendix 3) for the 1 x 1 m and 5 x 5 m plots. All plants were identified to the species level and their percentage cover, and that of bare ground, burnt material and brash were visually estimated (see Appendix 3 for detailed species lists and vegetation composition summaries). As there was a substantial bryophyte and lichen layer (henceforth referred to as moss layer as lichens and liverworts cover was very low) beneath the vascular plants, particularly pre-management, the overstorey (i.e. 'top view' vegetation seen when viewed from above) and understorey (i.e. covered by taller vegetation) were recorded separately. Overstorey cover of all species (and any brash and bare ground) totalled 100%, understorey cover was less than or equal to 100% and total cover therefore was between 100 and 200% as no plot had more than two layers of vegetation. Brash and bare ground cover was included in the overstorey layer (i.e. top view) where living vegetation was absent. However, in the understorey layer, brash was only recorded if it represented whole dead plants (i.e. litter was not included) and bare ground was not recorded if vascular plants were so dense that they formed both over- and understorey. A 1 m^2 quadrat split into 25 equal squares was placed over each 1 x 1 m sub-plot during the surveys to aid estimation of percentage cover (see **Figure 30** for examples of surveys after the initial management of burning and mowing in 2013). The 1 m^2 quadrat was left in place during the survey of the 5 x 5 m plot to give guidance for percentage cover estimation. All surveys were conducted by the same three observers apart from 2016, when only one (i.e. a trained botanist) was the same.

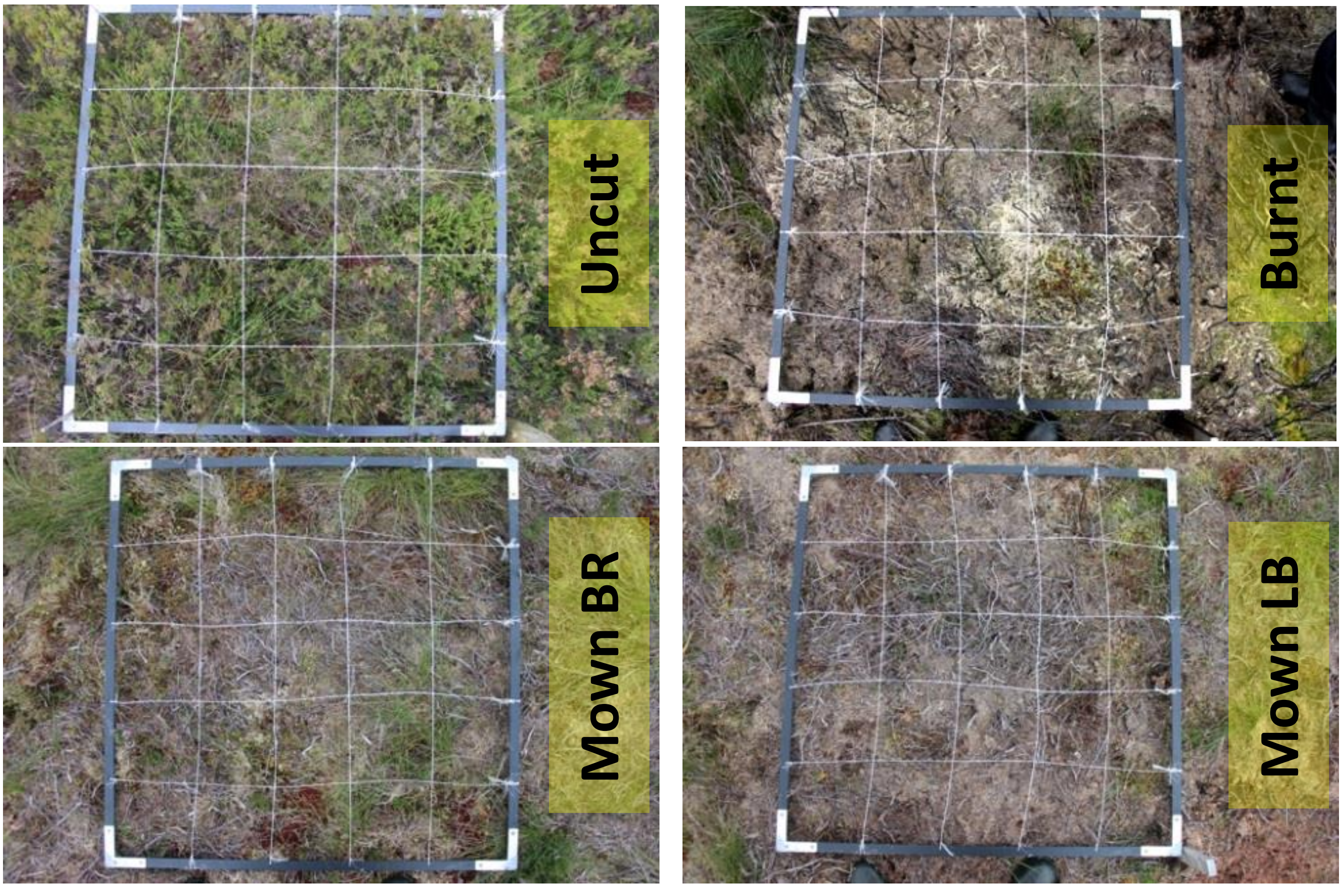


Fig. 30 Examples of representative plot conditions after management change in March 2013 (pictures were taken at Mossdale in October 2013). Note the burnt (*Sphagnum*) moss on the burnt plots, the sedge regrowth and intact (reddish colour) *Sphagnum* moss on the mown brash removal (BR) plots and the brash layer on the mown with left brash (LB) plots.

Changes in plant species composition, and separately in *Sphagnum* species composition, were analysed using redundancy analyses (see Section 4.2.6.4) to allow visual assessment of groupings over time and between managements. Moreover, (and in response to providing practitioner related outputs as requested by the project's

advisory group) percentage cover values from the field assessments of *Calluna*, *Eriophorum*, *Sphagnum*, non-*Sphagnum* mosses, bare ground, brash/dead/burnt material and all other species combined were compared to digital pictures taken at the same time (see **Figure 30** and **31** for examples and Section 4.2.6.1 for further details).

Fig. 31 Vegetation assessments in 2013 by manual assessment (**left**) and by taking digital images using a GoPro camera fixed on an extendable pole (**middle**, showing the method, and **right**, showing an example of the image taken at Mossdale). Manual and photo surveys were performed at both the 1x1 m and 5x5 m scales.

Calluna height was measured in five places on each 1 x 1 m and 5 x 5 m plot during the vegetation surveys and was averaged at both scales for each plot in each year. *Calluna* heights less than, or equal to, 3 cm were classed as having germinated from seed within the past year due to the observation that most of the regenerating shoots grew from woody stems at least 3-4 cm tall. *Calluna* plants were cut from the 29 cm diameter circles used for net ecosystem exchange (NEE) measurements (see Section 4.2.14) before mowing and burning were carried out to obtain pre-management *Calluna* measurements and plants were cut from similar sized areas in 2015 for post-management measurements (for dates see **Table A3.1** in Appendix 3). These samples enabled detailed assessment of heather cover (from pictures taken from above), volume, biomass (woody versus leafy) and leaf area which allowed upscaling heather data for these variables to the plot scale (see previous Section 4.2.5.2).

National vegetation classification (NVC) categories were determined for each management type at each site in 2012 and 2015 (pre- and post-management) using MAVIS software (DART Computing & Smart, 2014). Overall, the MAVIS software classified all plots at all sites in 2012 and 2015 as the NVC category M19a, which is the *Erica tetralix* sub-community of the *Calluna vulgaris – Eriophorum vaginatum* blanket mire community.

4.2.6.1 *Photo assessment comparison*

Assessment of photographs (see **Figure 31**) was compared to the ground level surveys for the major vegetation groups (see **Table 7**) for all 2012 to 2015 records. There were 84 photos (of a total of 570 with some form of moss) for which *Sphagnum* and non-*Sphagnum* moss could not be confidently distinguished, meaning moss was unclassified in less than 15% of the photos. Although field and photo survey percentage cover estimates appeared similar (**Table 7**), there was a significant difference (based on a paired Wilcoxon signed rank test; the reported V-values correspond to the sum of ranks assigned to the differences with positive sign and is similar to F-values.) between them (V_{2393} = 1509200, p = 0.022). When the data were split by the plot size, the percentage cover estimates between field and photo surveys were significantly different for the 5 x 5 m plots (V_{1138} = 354210, p = 0.006) but not for the 1 x 1 m plots (V_{1255} = 404060, p = 0.436). There was a significant difference between field and photo estimates for all species groups (p< 0.02 for all) but neither estimate was consistently higher for all categories (**Table 7**). Overall, there was a tendency for groups with low percentage covers to be underestimated in the photos, as plants were too small to see, and groups with higher percentage cover to be underestimated in the field, due to the viewing angle of the observer; this effect increased with increasing survey area. Moreover, non-*Sphagnum* moss was sometimes difficult to distinguish (on the camera pictures) from bare ground.

Table 7 Mean percentage cover (during 2012-2015) of *Calluna*, *Eriophorum*, *Sphagnum*, non-*Sphagnum* mosses, bare ground, brash/dead/burnt material and other species determined from the field and photo surveys split by plot size.

Species group	1 x 1 m field survey	1 x 1 m photo survey	5 x 5 m field survey	5 x 5 m photo survey
Calluna vulgaris	26.7	27.2	15.7	15.1
Eriophorum spp.	22.4	23.5	29.6	30.8
Sphagnum spp.	6.0	5.2	5.1	4.0
Non-*Sphagnum* moss	20.1	16.2	19.9	16.9
Brash/dead/burnt	2.9	2.6	2.6	2.9
Bare ground	21.1	24.5	26.1	29.9
Other spp.	1.0	0.7	1.2	0.3

Given that both methods involved surveying by eye, it is likely that human error contributed to some of the differences between field and photo estimates. However, only the 5 x 5 m plots differed significantly in their cover estimates, mainly due to the picture resolution being too low to accurately distinguish between small species, species at low cover and non-*Sphagnum* moss versus bare ground. Given that the estimates made of cover in the 1 x 1 m photo surveys were similar to those made by human recorders in the field, there is real potential to apply this method to engage landowners in habitat monitoring as well as producing scientifically meaningful data. However, a photo survey only allows assessment of the overstorey cover, and in particular does not allow the full capture of the understorey moss layer (i.e. bryophyte and lichen vegetation) or smaller or rare understorey species or identifying specific *Sphagnum* species (i.e. as indicator species for habitat condition). Therefore, only the detailed ground assessment was used in the vegetation data analysis reported below.

4.2.6.2 *Sphagnum pellet additions*

Sphagnum was added by means of pellets (BeadaMoss, Micropropagation Services, Loughborough, UK) on the 25th (Nidderdale and Mossdale) and 26th (Whitendale) March 2014 (**Figure 32**). This was done after management had taken place in order to “sow” into exposed areas. The BeadaMoss pellets consisted of fragments or “propagules” of a single *Sphagnum* species encased in a water gel coating, which was intended to protect the *Sphagnum* fragments from desiccation until the moss had established. Pellets containing *S. capillifolium, S. papillosum and S. palustre* were used in equal amounts. For each BR+Sp and LB+Sp plot, a quadrat was placed over the 1 x 1 m corner sub-plot which subdivided it into 25 equal squares. Thirty-three BeadaMoss pellets of each species were added to the 1 x 1 m sub-plot (i.e. 99 pellets m^{-2} in total; compared to a by BeadaMoss recommended application rate of 20 pellets m^{-2} for larger areas and 40-60 pellets m^{-2} on small selected plot plantings), with each quadrat square receiving at least one pellet of each species. The remainder of each 5 x 5 m plot was divided into 1 m strips and the *Sphagnum* pellets were applied to each strip separately, at a rate of approximately 100 pellets m^{-2} (i.e. the same application rate as per sub-plot), by shaking a pot from side to side to prevent pellets from clumping and sticking together. For the purposes of vegetation surveys, an additional FI (burnt) plot was established, with *Sphagnum* propagules added in the same manner as to the BR+Sp and LB+Sp plots. Additionally, in order to assess whether there was a specific density at which adding more *Sphagnum* propagules produced no further increase in *Sphagnum* cover, 1 x 1 m squares were established at Whitendale outside the four burnt (+*Sphagnum*) and a further four mown with left brash (LB+Sp) plots; 200, 400 or 600 beads were allocated to an area of 1 m^2 outside each plot, increasing in application rates (**Figure 32**) from top to bottom along the side of the plot (starting next to the 1 x 1 m survey square).

Fig. 32 Example of manual *Sphagnum* pellet distribution (26/03/13) across individual 1x1 m plots (**top left**) and across the 5x5 m plots (scattered volume). The pictures on the **top right** show the detailed allocation of pellets per 20 x 20 cm grid square (yellow arrows indicate pellet locations) within each 1x1 m area, whereas the three pictures (**bottom row**) show a close-up of the three densities applied at Whitendale outside selected mown and burnt plots (increasing from left to right).

There was no visual evidence of successful establishment of Beadamoss pellets until 2016, when two plots at Whitendale grew *S. capillifolium*, where previously none had been present. The relatively slow appearance (or failure) of *Sphagnum* may be due to the fact that the *Sphagnum* propagules were only added to relevant plots in 2014, whereas burning and mowing was carried out in 2013. However, recent data (personal communication Neal Wright, BeadaMoss) suggest that around 6 years are needed for pellet establishment in upland moorlands. So far a graphical analysis over time for the average cover of the added *Sphagnum* species did not show any positive application effect for any of the three species (see Fig. A3.3a and A3.3b in Appendix 3 for the 1 x 1 m and the 5 x 5 m plots). Therefore, for the purposes of analysis, any plots to which *Sphagnum* was added were combined with the other plots of the same management (i.e. FI+Sp and FI, LB+Sp and LB, and BR+Sp and BR plots). This resulted in four management groups - burning (FI), mowing with the brash left (LB), mowing with brash removed (BR) and the uncut comparison (DN) - meaning that for vegetation surveys, across the 3 sites, and with 4 replicate plots at each site, there were 12 DN replicates and 24 FI, LB and BR replicates (i.e. the latter three all had a ±Sp pellet addition).

4.2.6.3 *Heather growth and vegetation dynamics*

Heather growth:

Calluna height was different between managements in the post-management (but not pre-management) period when pooled across all sites (as there were no significant site x management effects) on both the 1 x 1 m sub-plots (**Figure 33**; top; χ^2_3 = 128.88, p< 0.001) and 5 x 5 m plots (χ^2_3 = 138.49, p< 0.001). On both plot sizes, as expected, uncut (DN) plots consistently had significantly taller *Calluna* than other management plots (p< 0.001 for all), being on average about 35 cm tall. On 5 x 5 m plots, *Calluna* was also significantly shorter on burnt (FI) plots than on mown LB and BR plots (p< 0.001 for all) in 2013, 2014 and 2015, but not in 2016, when *Calluna* reached average heights of more than 15 cm (**Figure 33**; top) on all 5 x 5 m management intervention plots.

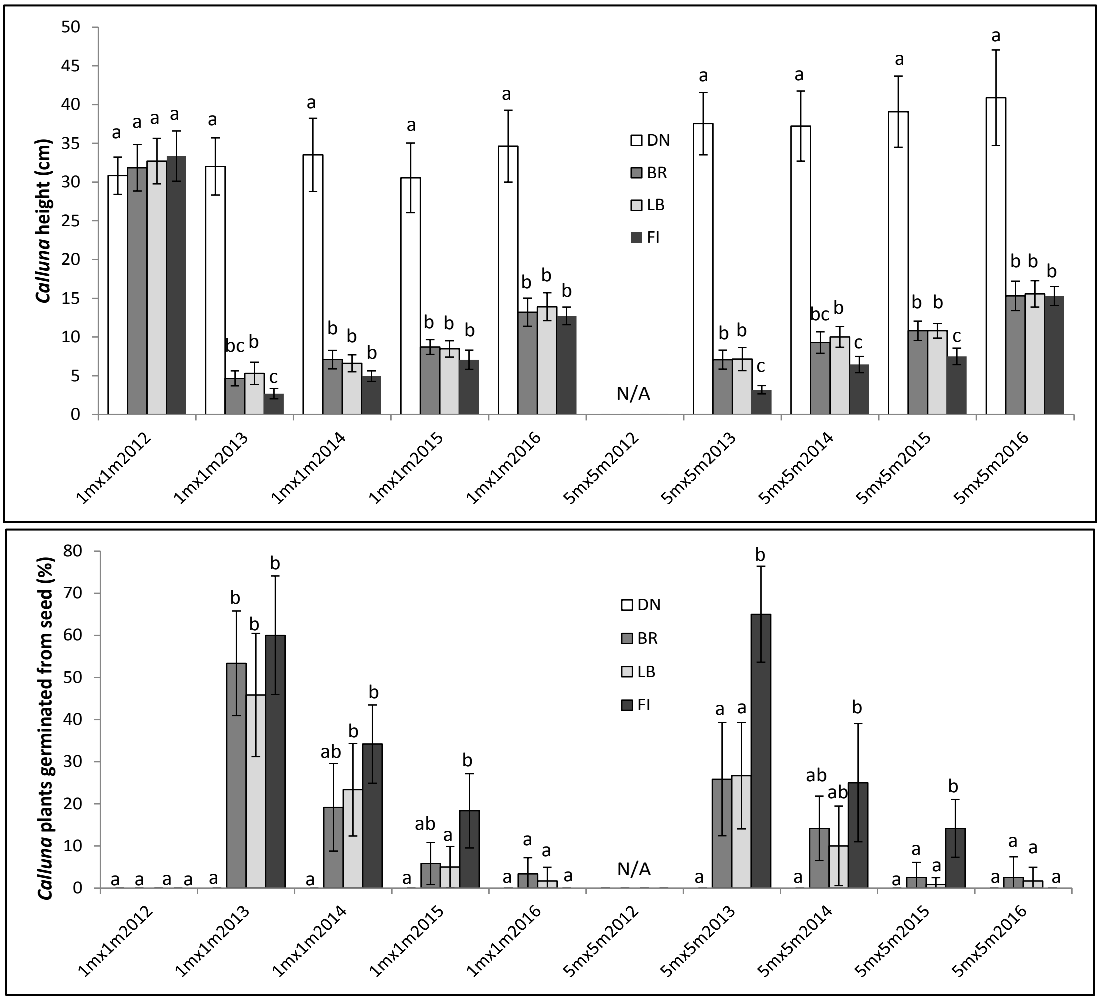


Fig. 33 Average (± 95%) *Calluna* heights (**top**) and percentage of *Calluna* germinated from seed (i.e. plants were <3 cm tall) (**bottom**) pooled across all three sites for the 1x1 m and 5x5 m plots in each year (2012: pre-management period had no 5x5 m data) for each management (DN is uncut, FI is burnt, BR is mown with the brash removed and LB is mown with brash left). Different letters within each year and survey area indicate significant differences (at p<0.05) between the managements.

No *Calluna* was classed as germinating from seed in 2012, nor did any germinate on uncut plots in any year (i.e. all recorded plants were >4 cm). The percentage of *Calluna* plants which were classed as germinated from seed was highest in 2013 and decreased thereafter (**Figure 33**; bottom). There were significant post-management differences in percentage of germinated *Calluna* plants measured on different managements on both the 1 x 1 m sub-plots (χ^2_3 = 40.04, p< 0.001) and 5 x 5 m plots (χ^2_3 = 42.9, p< 0.001). FI plots had generally higher germination than LB or BR plots over the period 2013-2015. This effect was significant on 5 x 5 m plots in 2013 and 2015, and on 1 x 1 m plots in 2015. However, there was no significant difference between management interventions in 2016, by when the percentage of germinated *Calluna* was very low, with *Calluna* reaching average regrowth heights of more than 15 cm (**Figure 33**; top) on all managed 5 x 5 m plots.

Dynamics in total vegetation cover over time:

To aid interpretation of the overall effects of the three main managements (uncut, mown and burnt), overstorey only vegetation and top view ground cover forms were grouped and plotted against time for 5 x 5 m plots (**Figure 34a**), which revealed some apparently transient changes. For all groups apart from *Calluna* (i.e. for cotton-grass, *Sphagnum*, non-*Sphagnum* mosses, other species and bare/brash/dead material) cover increased in 2013, after management change, on burnt and mown areas, probably due to the large decrease in *Calluna* cover. Despite initial differences between burnt and mown areas in cover of most groups in the year of management intervention, most of these differences had lessened, or in the case of *Calluna*, *Sphagnum* and bare/brash/burnt largely disappeared, by 2016, highlighting the importance of long-term monitoring in relation to evidence on likely long-term shifts in vegetation composition. However, as **Figure 34a** shows only percentage cover of vegetation as seen from above, much of change over time in *Sphagnum*, non-*Sphagnum* moss and bare/brash/burnt cover can be attributed to the regrowth of the larger plants such as *Calluna* and cotton-grass which obscured the visibility of understorey species from the top view.

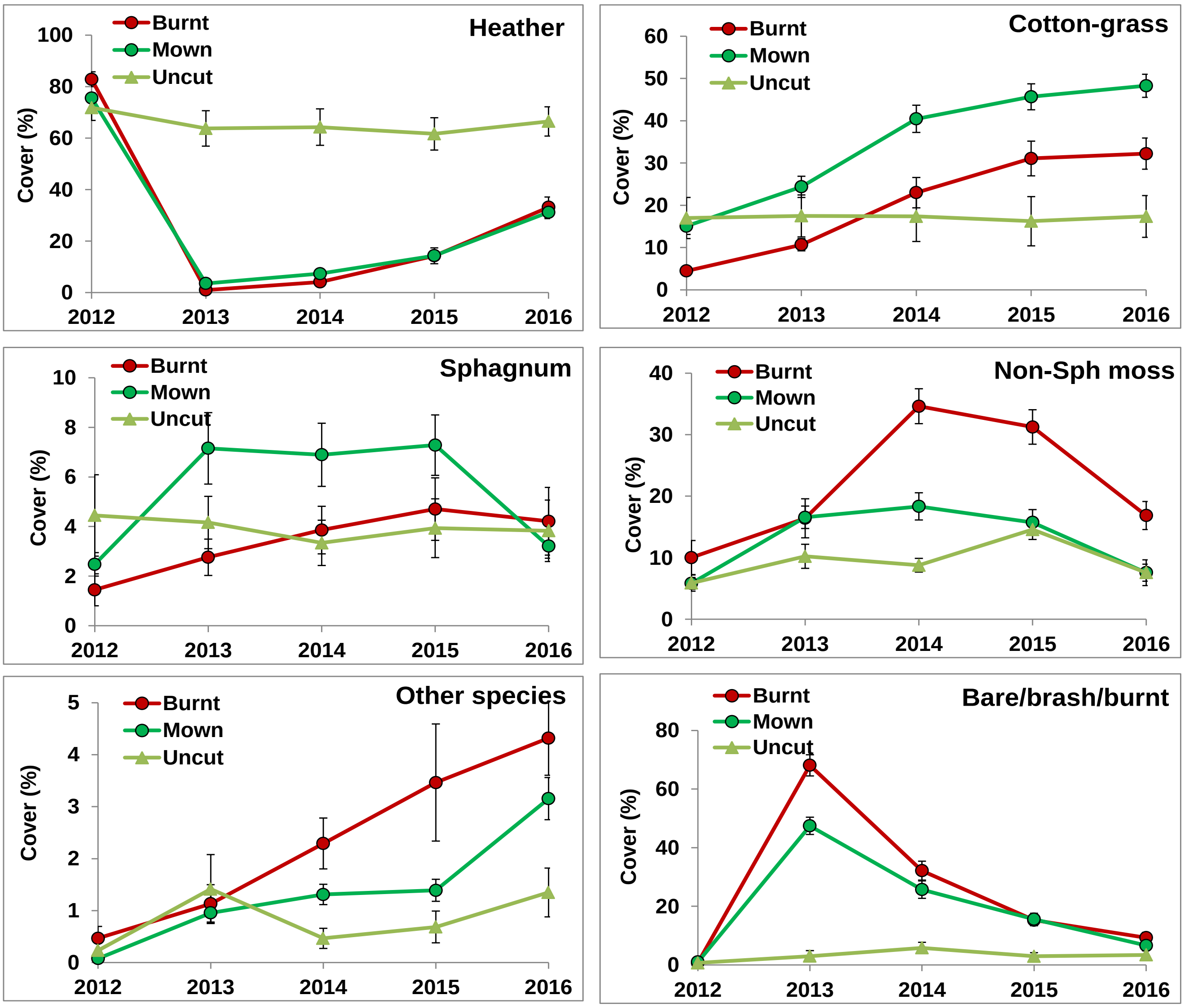


Fig. 34a Average (± 95% confidence interval) percentage cover of overstorey (or exposed 'top view') vegetation only for individual key species, groups and non-vegetated ground across all three sites as assessed by 5x5 m ground surveys during the pre- (2012) and the post-management (2013-2016) periods for each of the main management treatments (Burnt = FI; Mown = combined mowing treatments BR & LB, which were statistically non-different; Uncut = DN).

Figure 34b shows the total percentage cover for the 5 x 5 m plots (i.e. combined 'top view' and understorey), which revealed a similar picture as for the top view (**Figure 34a**) for heather, cotton-grass, other species and

bare/brash/burnt. However, the change over time in *Sphagnum* and non-*Sphagnum* moss was very different if the hidden moss component by re-growth of the larger plants such as *Calluna* and cotton-grass was taken into account (i.e. total cover). *Sphagnum* cover increased considerably on the mown plots in 2016, whereas on burnt plots it increased only very slowly throughout the post-management period. Whereas non-*Sphagnum* moss cover remained low on mown plots, on the burnt plots it increased over time after an initial decline and became more similar to the uncut (but previously burnt) plots where it also increased post-treatment. Moreover, total cotton-grass cover differed less between management treatments at the start in 2012 than for the top view (**Figure 34a**). Therefore, cotton-grass, *Sphagnum* and non-*Sphagnum* moss cover together revealed the possible onset of an overall different trajectory, with burnt plots remaining similar to uncut plots but mown plots supporting more 'active' bog species (see also Appendix 3a for a more detailed species-level/ecological vegetation assessment). It is important to note the value of the total cover ground-level assessment (**Figure 34b**) compared to the 'top view' only cover (**Figure 34a**) as both aerial photography (e.g. drone surveys) and remote sensing (e.g. satellite imagery) assessments would miss these crucial differences. There were differences in cover between sites (see also Appendix 3a), although the broad trends at each site were comparable (see **Figure A3.2** in Appendix 3); these differences between sites are assessed in more detail for the 2016 data below.

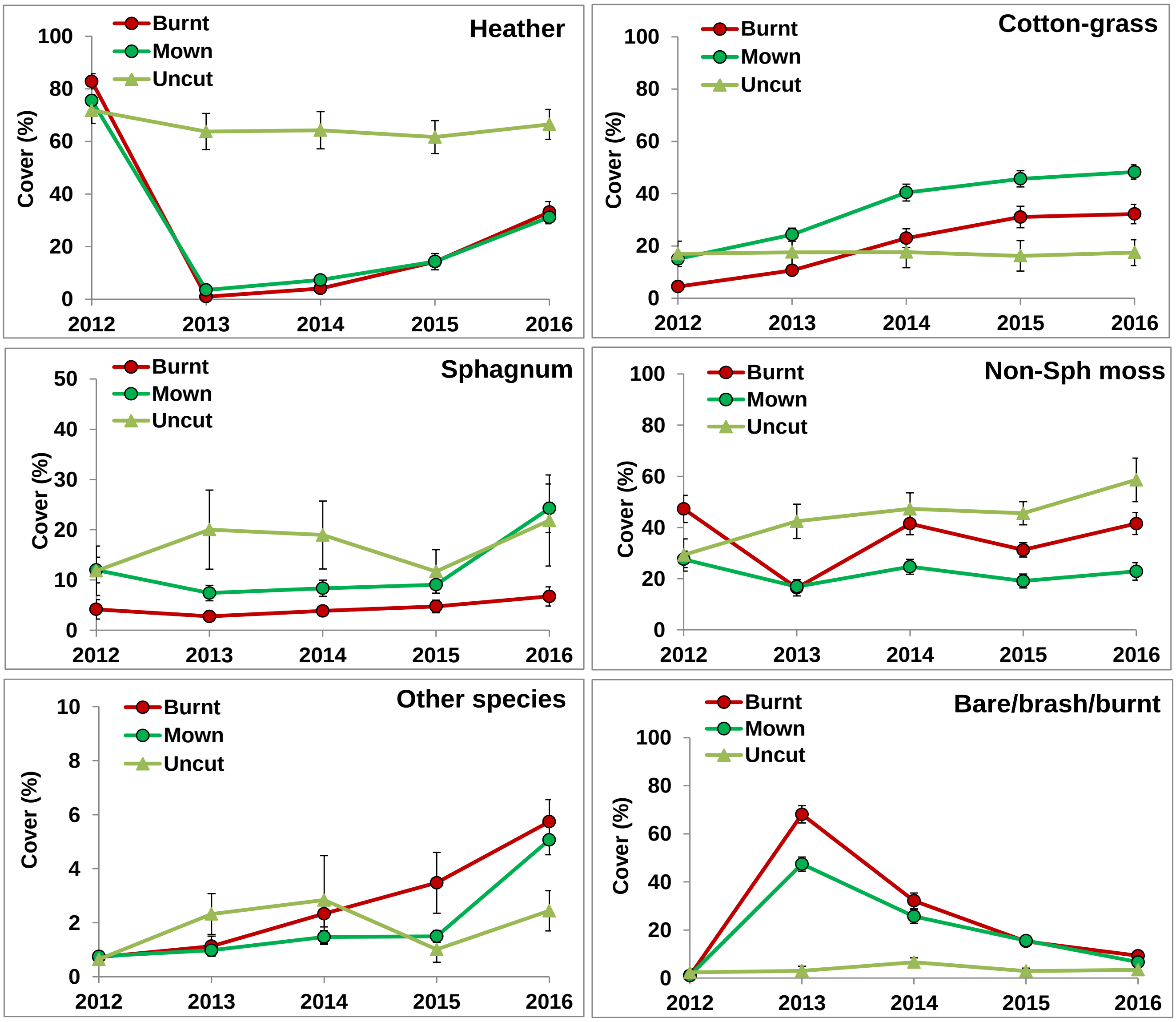


Fig. 34b Average (± 95% confidence interval) percentage of total vegetation cover (combined overstorey and understorey) for individual key species, groups and non-vegetated ground across all three sites as assessed by 5x5 m ground surveys during the pre- (2012) and the post-management (2013-2016) periods for each of the main management treatments (Burnt = FI; Mown = combined mowing treatments BR & LB, which were statistically non-different; Uncut = DN).

Change in vegetation cover pre- versus post-management:

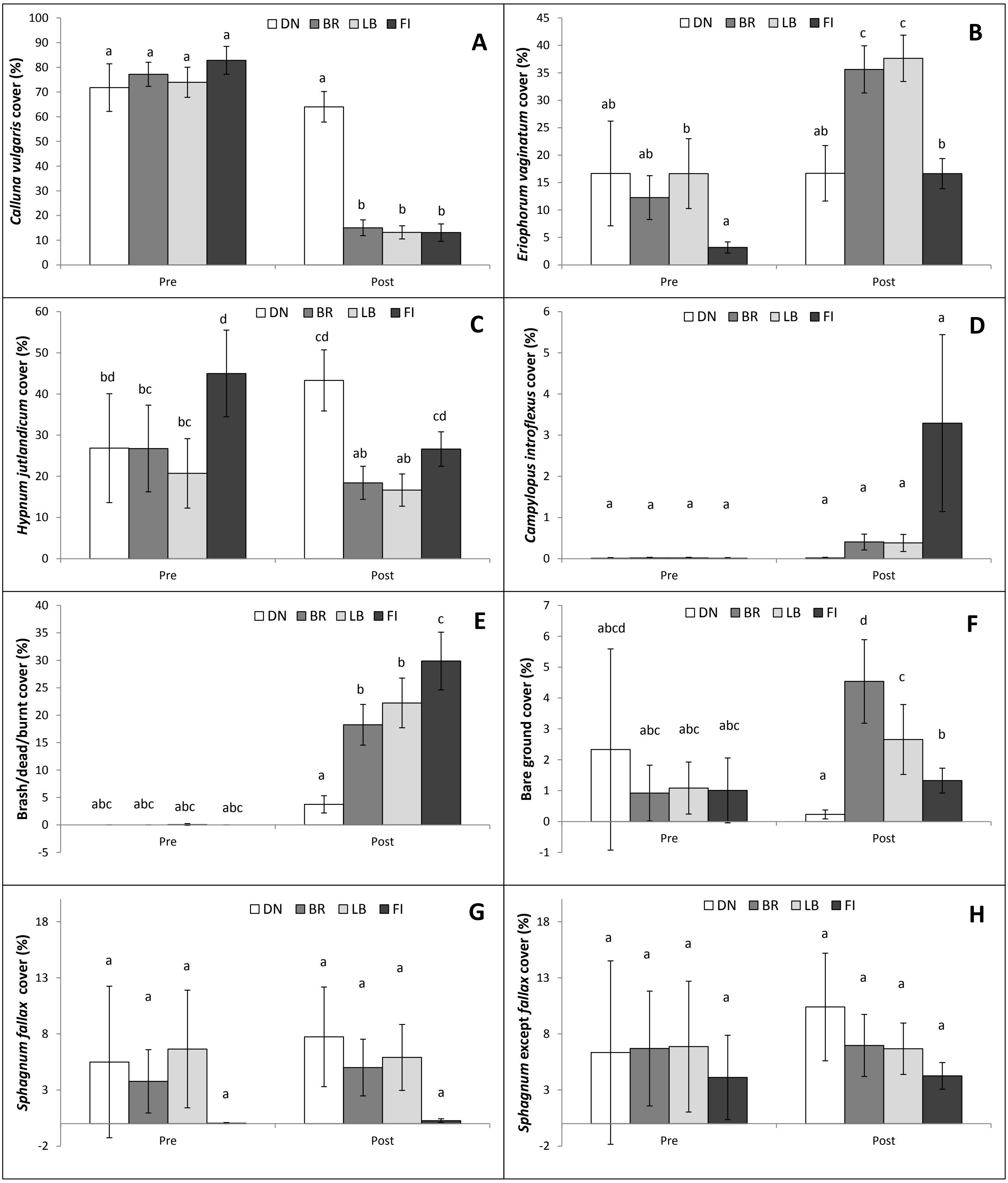


Fig. 35 Mean (± 95% confidence interval) percentage total cover (i.e. including the understory) of **A**) *Calluna vulgaris*, **B**) *Eriophorum vaginatum*, **C**) *Hypnum jutlandicum*, **D**) *Campylopus introflexus*, **E**) brash/dead/burnt material, **F**) bare ground, **G**) *Sphagnum* fallax, and **H**) all *Sphagnum* spp. (except *S. fallax*) for each management group in the pre- (2012) and post-management (2013-2016) periods for the 5x5 m plots. Management codes are DN (uncut), FI (burnt), BR (mown brash removed) and LB (mown brash left). Different lower case letters within each time period indicate significant differences between managements. Please note the different y-axis scales.

Formal statistical analysis of these differences in cover took two forms. Firstly, data for total cover for major individual species and species group was analysed for the combined post-management periods, and also for the pre-management period, to ensure any plot differences that were present before management intervention were identified. This analysis was based on data pooled across the three sites, since no significant site x management interactions were found. Secondly, data for the latest year (2016) was analysed, in this case also considering differences between sites, to assess the position four years after management intervention.

When data were pooled across the three sites, analysis of vegetation total percentage cover (see **Figure 35** above) revealed that *E. vaginatum* (one of the two cotton-grass species) showed significant differences in the pre-management period (**Figure 35B**), with mown LB (but not BR) plots having significantly higher cover than burnt (FI) plots ($p = 0.016$). Percentage cover of *E. vaginatum* increased significantly from the pre-management to post-management period on BR and LB plots ($p < 0.002$) and on FI plots ($p < 0.03$), although in the case of FI this was only to pre-management values of the non-FI plots; as expected, there was no change in cover on uncut (DN) plots ($p = 0.999$). This resulted in mown BR and LB plots having significantly higher *E. vaginatum* cover than DN and FI plots post-management (see **Table 8** for a statistical summary of overall significance of management and of specific management contrast post-intervention).

Table 8 Results from linear mixed effects models of interaction between management and time period (pre- and post-management: 2013-2016) for each vegetation species or group total cover on 5x5 m plots. For each model (for which stable models could be derived), the F value, the p value and the Satterthwaite approximation of the denominator degrees of freedom (df) are given (the numerator df = 3 for all models). Results of post-hoc tests for pairwise comparisons of the managements are also given, where DN is uncut, BR is brash removed, LB is left brash and FI is burnt. Only post-hoc results from the post-management period are shown as there were no significant differences pre-management, when all plots were uncut, except for *Eriophorum vaginatum* and *Hypnum jutlandicum* (see text for details). Significant values are shown in bold.

Species	F value	p value	df	DN to BR	DN to LB	DN to FI	BR to LB	BR to FI	LB to FI
Calluna vulgaris	26.82	**<0.001**	343	**<0.001**	**<0.001**	**<0.001**	0.937	0.090	0.697
Brash/dead/burnt material	10.65	**<0.001**	345	**<0.001**	**<0.001**	**<0.001**	0.577	**<0.001**	**0.001**
Hypnum jutlandicum	3.82	**0.0103**	342	**<0.001**	**<0.001**	0.391	0.976	**<0.001**	**<0.001**
Eriophorum vaginatum	7.38	**<0.001**	341	**<0.001**	**<0.001**	0.300	0.983	**<0.001**	**<0.001**
Eriophorum angustifolium	1.73	0.161	344	-	-	-	-	-	-
Sphagnum fallax	0.14	0.938	342	-	-	-	-	-	-
Bare ground	9.36	**<0.001**	343	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.045**
Campylopus introflexus	2.49	0.060	344	-	-	-	-	-	-
All *Sphagnum* spp.	0.11	0.951	342	-	-	-	-	-	-

As expected, DN plots had significantly higher *Calluna* cover and significantly lower cover of brash/dead/burnt material and bare ground compared to the other interventions after management (see **Figure 35A** and **Table 8**). There were no significant differences in *Calluna* cover between FI, LB and BR plots post-management, but there was significantly higher cover of brash/dead/burnt material on FI plots compared to BR and LB plots (**Figure 35E** and **Table 8**) whereas BR plots had significantly more bare ground than LB and FI plots (see **Figure 35F** and **Table 8**), reflecting exposing bare ground by raking off the brash. *Hypnum jutlandicum* showed significantly higher cover after burning than LB and BR plots (**Figure 35C** and **Table 8**), but, as was the case for *E. vaginatum*, this was reflecting pre-management differences. In contrast, *Sphagnum* species (**Figure 35G-H**) showed no significant management effect for total cover over the entire post-management period (**Table 8**). *Campylopus introflexus* recolonization after management was higher on burnt than mown plots, but this effect was not significant over the entire post period (**Figure 35D** and **Table 8**).

For a more in-depth ecological vegetation composition assessment (i.e. species level) in relation to habitat condition, and specifically cover of 'active' bog indicator species see Appendix 3a. This Appendix also provides an additional comparison between sites, sub-catchments and management pre- versus post-management and over time in general for both survey scales (i.e. 1 x 1 m and 5 x 5 m).

Change in vegetation cover immediately post-management:

The response to management of total cover (including understorey vegetation) of the main species cover groups was also most pronounced immediately after management (except for the *Sphagnum* and non-*Sphagnum* moss cover, see **Figure 34b**).

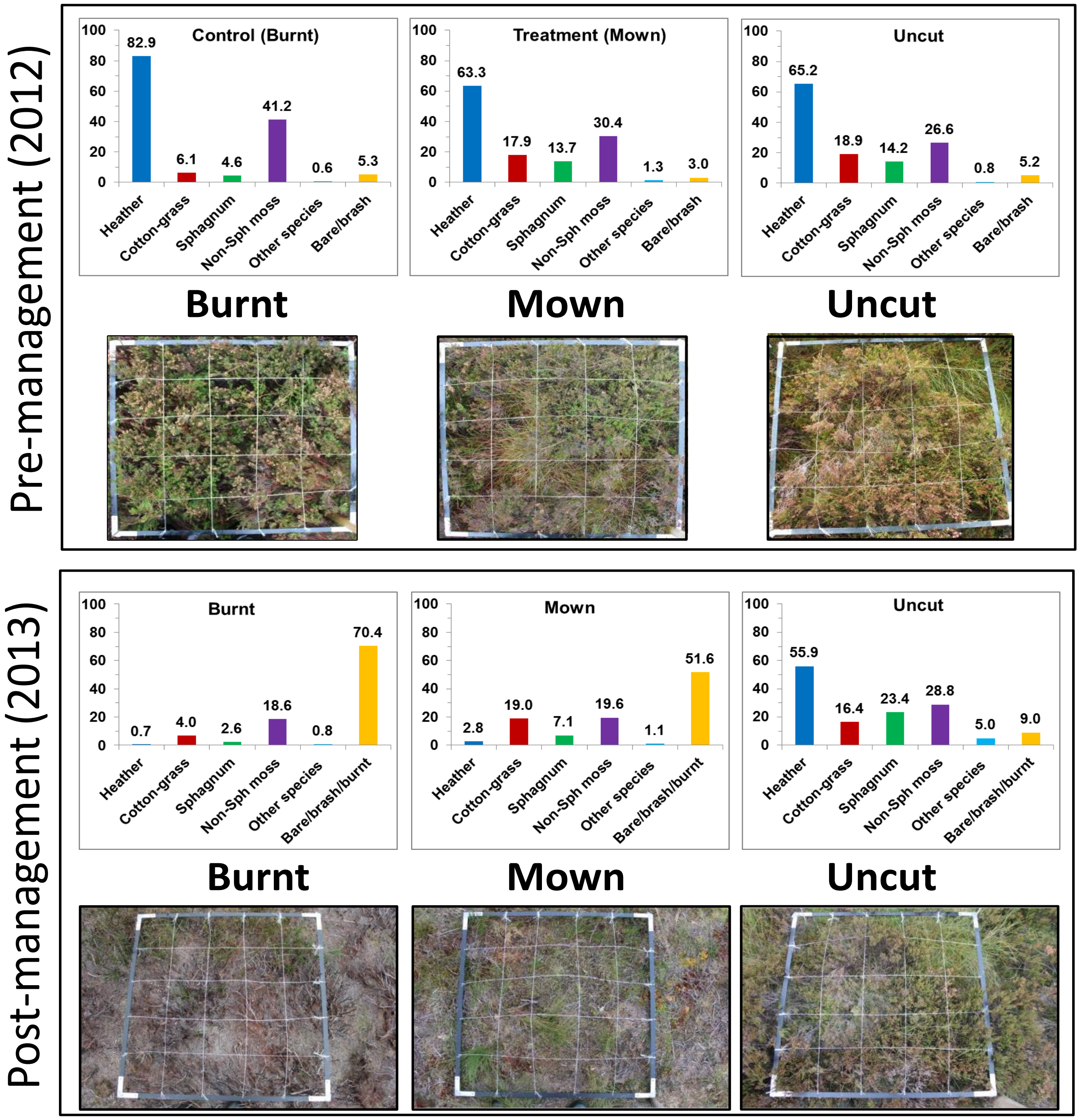


Fig. 36 Mean total percentage cover (including understory, therefore adding up to more than 100%) of the 1x1 m monitoring plots combined across all three sites and mowing treatments, for 2012 (pre-management; **top**) and 2013 (post-management, **bottom**). For clarity percentages are displayed without error bars for the main species, vegetation types and bare ground.

When total cover from 1 x 1 m plots (**Figure 36**), which was similar to the 5 x 5 m plots (cf. **Figure 34a,b**), was compared between managements averaged across sites, there were clear management impacts in 2013, the first post-treatment year. Bare/brash/burnt ground cover was ~20% greater and cotton-grass cover was ~10% less under burning compared to the combined mowing management (although this was already the case for cotton-grass before management for the overstorey cover; **Figure 34a** and **Figure 36**); yet whilst *Calluna* cover was

reduced to less than 5% under both managements, living *Sphagnum* cover (dead cover was included in bare/brash) was about 5% less on burnt than on the combined mown management plots. Cover of other moss species combined was initially (i.e. 2013) reduced by both management interventions to less than uncut plots. The increase in *Sphagnum* cover on mown plots for the 'top view' (**Figure 34a**) may also reflect spreading of fragments by the mowing process, while the lack of change between pre- and post-period (**Figure 35**) on burnt plots may indicate that there was limited fire damage. However, pre-management there was already nearly three times higher *Sphagnum* cover on mown than burnt plots (across all three sites). Moreover, *Sphagnum* cover also doubled on uncut plots between the pre- and post-management period, likely indicating an opening up response of previously hidden or shaded ground vegetation, perhaps as a result of heather die back due to desiccation after harsh frosts in 2012/13 (see **Figure 10**) with a lack of protecting snow cover, particularly at the wetter sites Mossdale and Whitendale. Damage to *Calluna*, such as by heather beetle, can be expected more readily when the plant is under stress from, for example, rising water tables (i.e. Gimingham, 1960), as was achieved by mowing; additional years should reveal if this will lead to a general decline in *Calluna* cover as indicated by frost and heather beetle damage at the two wetter sites.

As for the overstorey data (cf. **Figure 34a**), the total cover of the major species and groups under management interventions became more similar to the uncut plots at all three sites by 2016 (cf. **Figure 37**). Detailed statistical analysis of effects of management and site used total cover data, including the understorey. Comparison between sites revealed that Nidderdale had highest *Calluna* cover across all plots post-management (significant interaction between site and period; $F_{2,\ 372} = 6.53$, $p = 0.002$), but there was no significant interaction between management, period and site ($F_{6,\ 372} = 1.55$, $p = 0.16$), despite *Calluna* cover at Whitendale almost halving on DN plots following frost damage in winter 2012/13. *E. vaginatum* cover also differed significantly between sites ($F_{2,\ 374} = 21.55$, $p < 0.001$), with Whitendale having the highest cover, but there was no significant interaction between management, period and site ($F_{6,\ 374} = 0.70$, $p = 0.65$). Likewise, *Sphagnum* cover was significantly different between sites ($F_{2,\ 374} = 296.57$, $p < 0.001$), with Mossdale having by far the most *Sphagnum* cover, but without any significant interaction between management, period and site ($F_{6,\ 374} = 0.47$, $p = 0.83$). For a further ecological assessment see the additional analysis provided in Appendix 3a.

Differences in vegetation cover in 2016:

Notably, large differences (but plot variability was high, cf. **Figure 35**) between the two management interventions remained in 2016 for cotton-grass and mosses, with burnt plots having more than three-fold less *Sphagnum* and nearly double non-*Sphagnum* moss cover than mown plots, on all three sites (**Figure 37**). Importantly, by 2016 (**Figure 37**) mown plots were more similar to uncut plots in terms of higher *Sphagnum* cover, whereas the burnt plots had more similar cover of cotton-grass and non-*Sphagnum* moss (particularly due to *Hypnum jutlandicum*) to the uncut plots than to the mown plots. However, the direction of long-term change beyond the four years of post-management monitoring remains uncertain, and will reflect different species' growth rates and adaptations to changes in shade and moisture as vegetation reaches maturity.

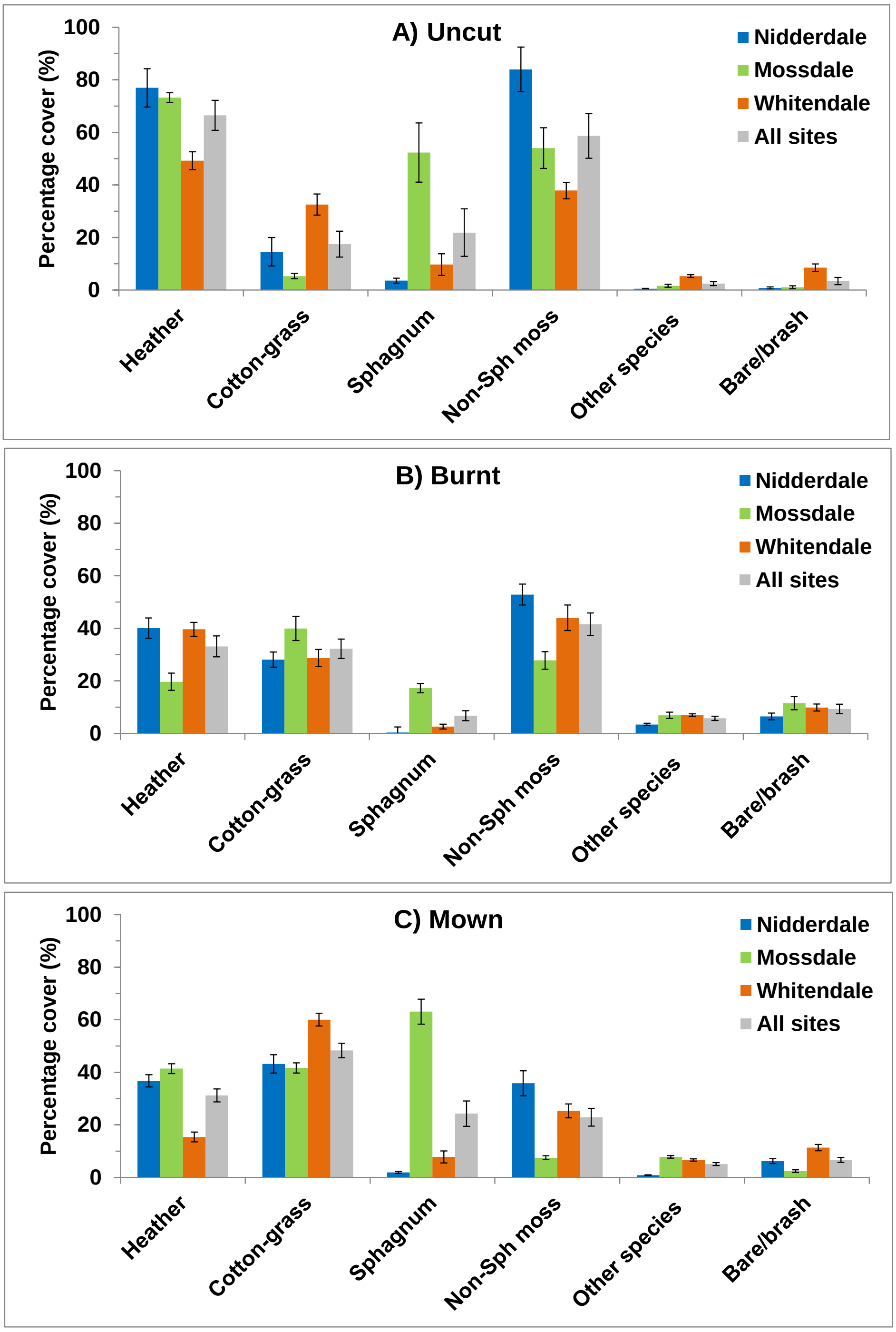


Fig. 37 Mean (± 95% confidence interval) total percentage cover (including understory, therefore adding up to more than 100%) of the 5x5 m monitoring plots in 2016 at the individual sites and all sites combined (All sites), for: **A**) uncut (DN), **B**) burnt (FI) and **C**) mown (combined mowing treatments BR & LB) plots. Percentages are displayed for the main species, vegetation types and bare ground (including brash cover).

Differences in *Sphagnum* cover pre- versus post-management:

There were large differences in moss cover between the sites, with Mossdale having by far the most *Sphagnum* cover (**Figure 37** above and **38** below). When the data for all 5 x 5 m plots were combined at each site, there was an inverse relationship between the number of *Sphagnum* species and the total cover of *Sphagnum*. Mossdale had a much higher percentage cover (mean over the five years of 27.6%) than the other two sites, but only 8 species with two, *S. capillifolium* and *S. fallax*, overwhelmingly dominant, whereas Whitendale had lower cover (mean of 3.5%) with 9 species and Nidderdale had the lowest cover (mean of 1.5%) but 11 species (**Figure 38**).

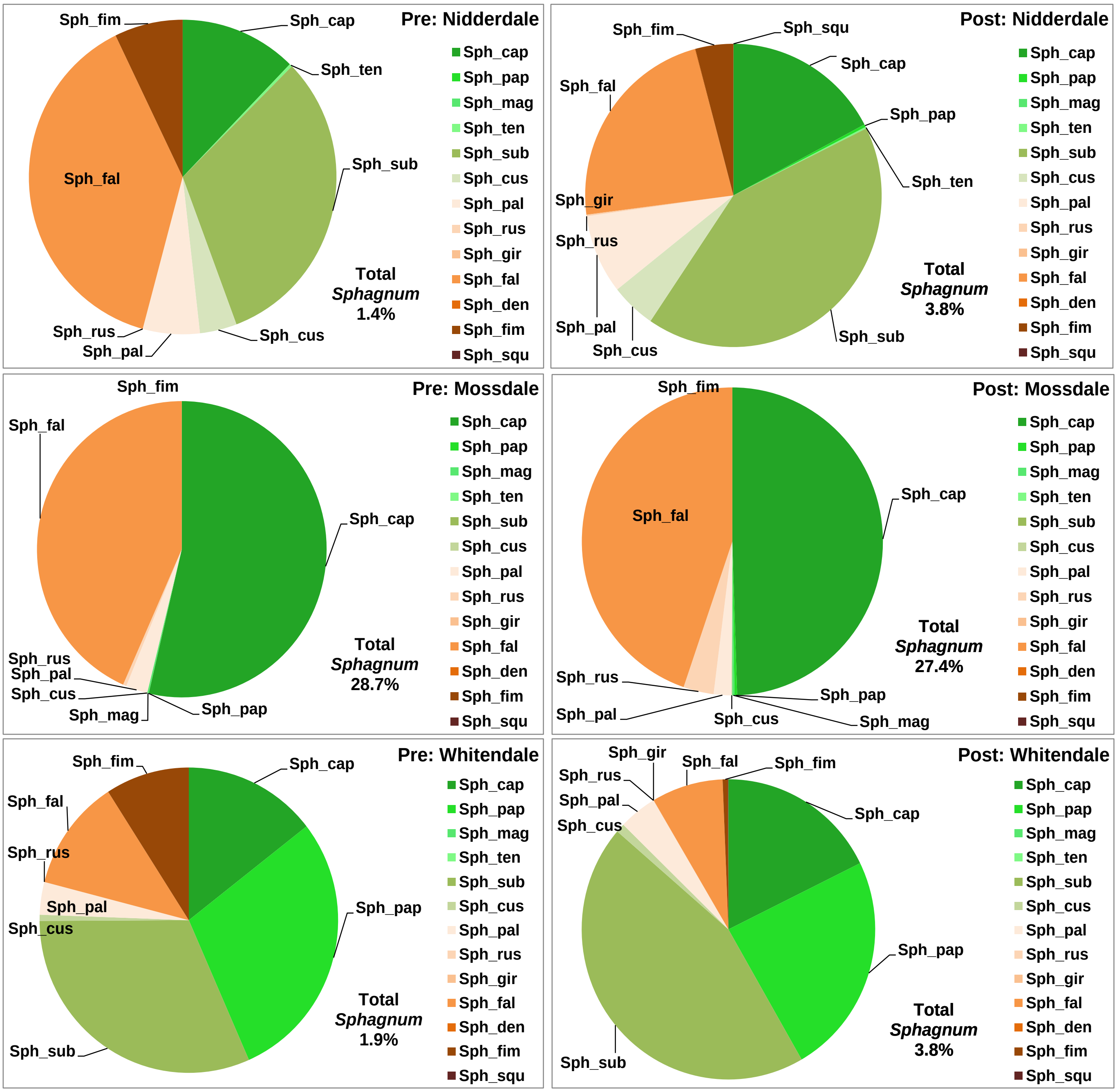
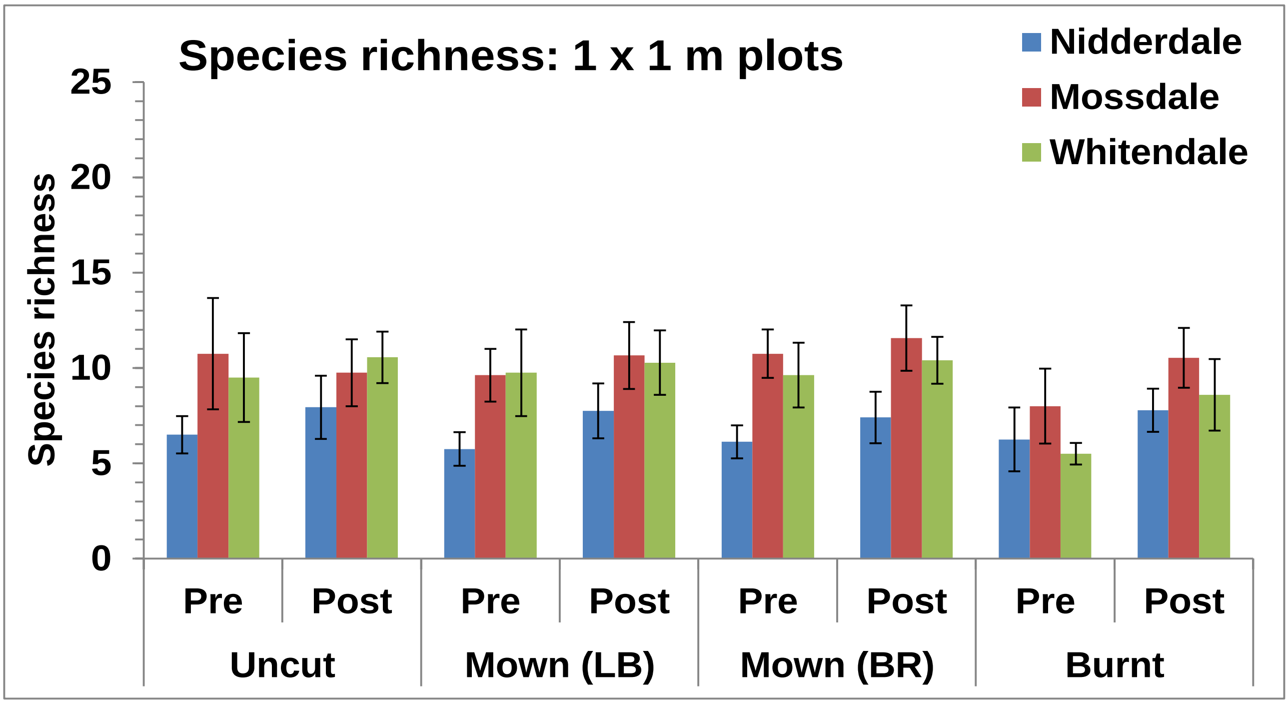


Fig. 38 The percentage of total cover of *Sphagnum* contributed by each *Sphagnum* species for each site during the pre- (left-hand diagram) and post-management (right-hand diagram) periods for all 5x5 m plots combined (n = 24). Total *Sphagnum* cover varied between sites, and is given in the bottom right-hand corner of each diagram. Species abbreviations (**from top to bottom**) are: *S. capillifolium, S. papillosum, S. magellanicum, S. tenellum, S. subnitens, S. cuspidatum, S. palustre, S. fimbriatum, S. girgensohnii, S. fallax, S. denticulatum, S. russowii, S. squarrosum.* Any species not labelled on a chart indicates no record of that species during the period at that site. The order from *S. capillifolium* to *S. squarrosum* reflects an overall ecological gradient (which is open to debate) in terms of habitat indication from 'ombotrophic bog' (greens) to 'minerotrophic fen' (browns) with a threshold around *S. cuspidatum*, which is supported by The Handbook of European Sphagna (Daniels & Eddy): http://nora.nerc.ac.uk/id/eprint/5242/1/Handbook_Euro_Sphagna.pdf.

The difference between sites in *Sphagnum* cover, especially *S. capillifolium* as an indicator of ombotrophic bog conditions (see **Figure 38**), is important, particularly given the importance of this key group for bog structure and function, it may also be important in influencing cover of other species (e.g. due to its moisture regulation) and hence composition over time in response to the treatments. This difference in the cover of *Sphagnum* species (total of 13 species across all sites) is likely to be related to moisture, since Mossdale was the wettest site (MAP of 2030 mm) and Nidderdale the driest (MAP of 1590 mm), but it may also reflect less intensive past burning (see Section 4.3.3). The moisture theory is supported by Campeau & Rochefort (1996) who found that all except one of the *Sphagnum* species they investigated had greatly increased cover in wetter conditions. The difference in number of species may also be related to the type of habitat each species prefers. For example, *S. tenellum* is usually found on bare peaty ground or well drained wet heaths (Amphlett & Payne, 2010) and this species only occurred at Nidderdale (**Figure 38**), probably because the other sites were too wet. Additionally, Breeuwer et al. (2009) found that *S. cuspidatum* and *S. magellanicum* responded in opposite ways to changes in water table, perhaps explaining why both these species were only found in localised areas in the present study. However, *S. fallax* was found at all sites (**Figure 38**) and was the only *Sphagnum* species showing influence in the redundancy analysis (see below and **Figure 41**). This species inhabits a wide range of habitats (Amphlett & Payne, 2010) which may explain its distribution and why it appeared to be unaffected by management (**Table 8**). Due to the low cover of the other *Sphagnum* species other than *S. capillifolium*, it is possible that individual *Sphagnum* species did or may in future respond differently to the various managements, but that these differences were not detectable at such low cover and over such a short post-management period. For a further ecological assessment see the additional analysis provided in Appendix 3a.

Species richness and diversity:

In total, 79 plant species, including 13 different species of *Sphagnum*, were found across the three sites during the five survey years (see **Table A3.1** in Appendix 3). Some species were found only at one site during one year whereas others were present at all sites throughout. However, not all species are characteristic of blanket bog and a more ecological assessment (by R.A. Linsday; University of East London) is provided in Appendix 3a. The species richness (**Figure 39**) differed significantly between sites and survey scales ($F_{1,\ 776} = 736.32$, $p < 0.001$), with Nidderdale being species poor compared to the other two sites. In the post-management period, there were no significant differences in species richness between management treatments. Although there was a significant interaction between management and time period ($F_{3,\ 776} = 3.81$, $p = 0.009$), the only significant difference was during pre-management between BR and FI.



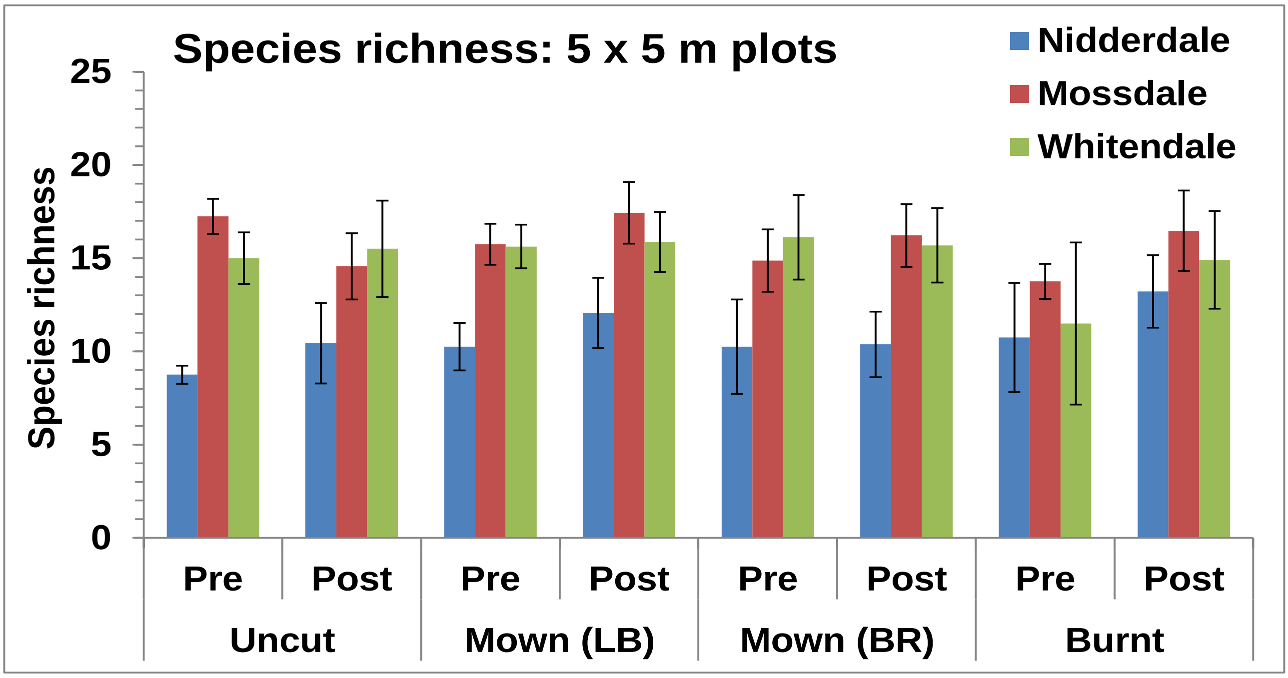


Fig. 39 The mean (± 95% confidence interval) species richness for the 1x1 (**top**) and the 5x5 m (**bottom**) plots for each management (DN = uncut; BR = mown brash removed; LB = mown brash left; FI = fire/burnt). For all treatments the combined ±*Sphagnum* pellet additions are shown.

The average number of species across all sites increased from 9.2 for the 1 x 1 m plots to 14.3 for the 5 x 5 m plots (**Figure 40**; top). However, the effective number of species (derived by taking the exponential of the Shannon's H diversity index) was not significantly different between the survey scales overall ($F_{1, 774} = 2.89$, $p = 0.089$; **Figure 40**; bottom), but was between the sites ($F_{2, 774} = 96.78$, $p < 0.001$), with Nidderdale having the lowest value. There was a significant interaction between management and time period ($F_{3, 774} = 6.52$, $p = 0.0002$; **Figure 40**); the effective number of species increased significantly from the pre-management to the post-management period on both burnt plot sizes ($p < 0.001$), with FI 1 x 1 m plots having significantly lower diversity than BR and LB plots ($p < 0.05$ for all) in the pre- but not the post-management period (**Figure 40**). There was no significant interaction between management, site and period ($F_{6, 749} = 0.94$, $p = 0.46$).

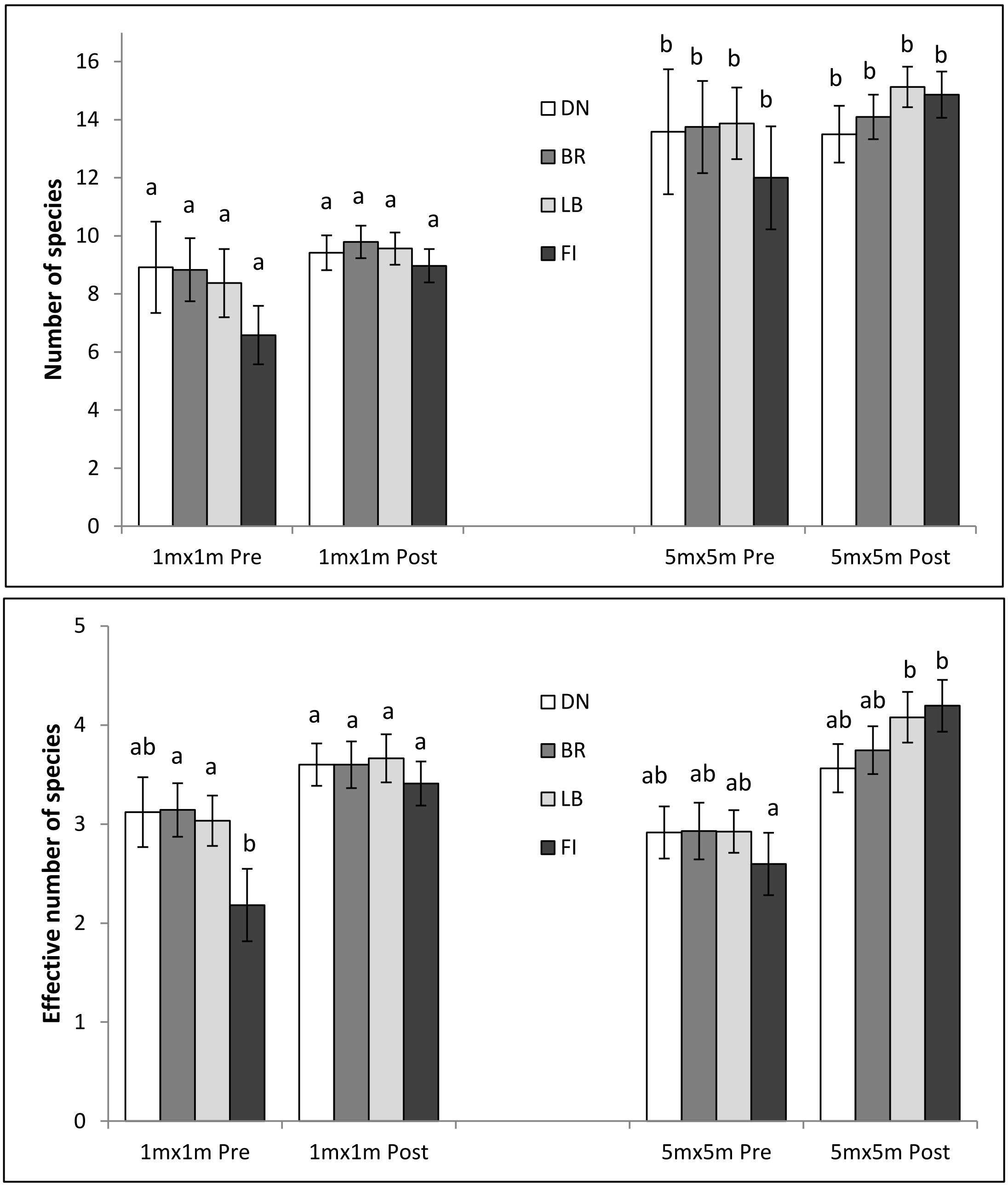


Fig. 40 The mean (± 95% confidence interval) number (**top**) and effective number (**bottom**; derived by taking the exponential of the Shannon H-indices) of species across all three sites for the different management plots (DN = uncut; BR = mown brash removed; LB = mown brash left; FI = fire/burnt) for either the 1x1 m or the 5x5 m monitoring plots. Different letters within each time period indicate significant differences between managements.

4.2.6.4 *Redundancy analyses*

To assist assessing management impacts in relation to vegetation and species groupings redundancy analysis (RDA) was performed. When management was used as the constraining variable in the RDA (**Figure 41** top; see next page), 14.6% of the overall variability of the Hellinger-transformed vegetation matrix during the post-management period (i.e. excluding 2012 data) was explained (adjusted $R^2 = 0.15$, $F_{3, 332} = 20.16$, $p < 0.001$).

The first axis separated plots which were dominated by brash/dead/burnt material from those dominated by *Calluna* (RDA1 axis of **Figure 41**). The second gradient separated observations with a high cover of *Hypnum jutlandicum* from those with a high cover of *Eriophorum vaginatum* and, to a lesser extent, *Sphagnum fallax* (RDA2 axis of **Figure 41**). The k-means partitioning satisfactorily divided the 336 observations into three groups ($F_{2, 332} = 89.64$, $p < 0.001$). These groupings, when plotted over the RDA graph, coincided closely with the centroids of the management groups, effectively creating a burnt (FI), mown (LB & BR) and uncut (DN) polygon. These polygons show that *Calluna* was the most closely associated species with the 'uncut polygon', *E. vaginatum* was with the 'mown polygon' and brash/dead/burnt material was with the 'burnt polygon'. Moreover, when the observations were coloured by their actual management (i.e. the coloured dots in **Figure 41**), 94% of the DN observations fitted into the 'unmanaged polygon' (or were in the small overlap), 74% of the FI plots fitted into the 'burnt polygon' (or were in the small overlap) and 61% of each of the BR and LB observations fitted into the 'mown polygon' (or were in the small overlap).

When (post) management was used as the constraining variable in the RDA but with calendar year, site and block partialled out (i.e. used as random effects variables) (**Figure 41**; bottom), management explained 15.3% of the variance of the vegetation matrix (adjusted $R^2 = 0.15$, $F_{3, 332} = 31.97$, $p < 0.001$). The axes separated out the plots similarly to those of the RDA which did not include year, site and block as random effects and the same species were associated with each polygon derived from the k-means partitioning. The observations coloured by management actually fitted these polygons better than in the RDA without random effects; 98% of DN observations fitted into the 'unmanaged polygon' (or the overlap between), 85% of FI plots fitted into the 'burnt polygon' (or overlap between) and 91% and 97% of the BR and LB observations respectively fitted into the 'mown polygon' (or the overlap). When year alone was used as the constraining variable in the RDA (plot not shown), it explained 13.2% of the variability of the vegetation matrix (adjusted $R^2 = 0.13$, $F_{3, 332} = 17.97$, $p < 0.001$). The first axis explained most of the variability (90.1%) and plots generally moved from brash/dead/burnt material in 2013, which were both on the far left, to *Calluna* and *Eriophorum* species on the right in 2016. Most uncut (DN) plots remained on the right-hand side for all years. This RDA indicated that the main change between years was that plots which had been burnt or mown regained vegetation, both becoming more similar to the uncut plots.

The eight most influential species/groups identified by the RDA were *Calluna*, brash/dead/burnt material, *H. jutlandicum*, *E. vaginatum*, *E. angustifolium*, *S. fallax*, bare ground and *Campylopus introflexus*. Additionally, all *Sphagnum* species (excluding *S. fallax*) were summed and analysed as a single group. Of these nine species/groups, total cover of *Calluna*, *E. vaginatum*, bare ground and brash/dead/burnt material showed significant interactions between management and time period (cf. **Table 8**); hence, there was consistency between the results of the redundancy analysis and those of the linear mixed modelling of cover (see previous Section 4.2.6.3).

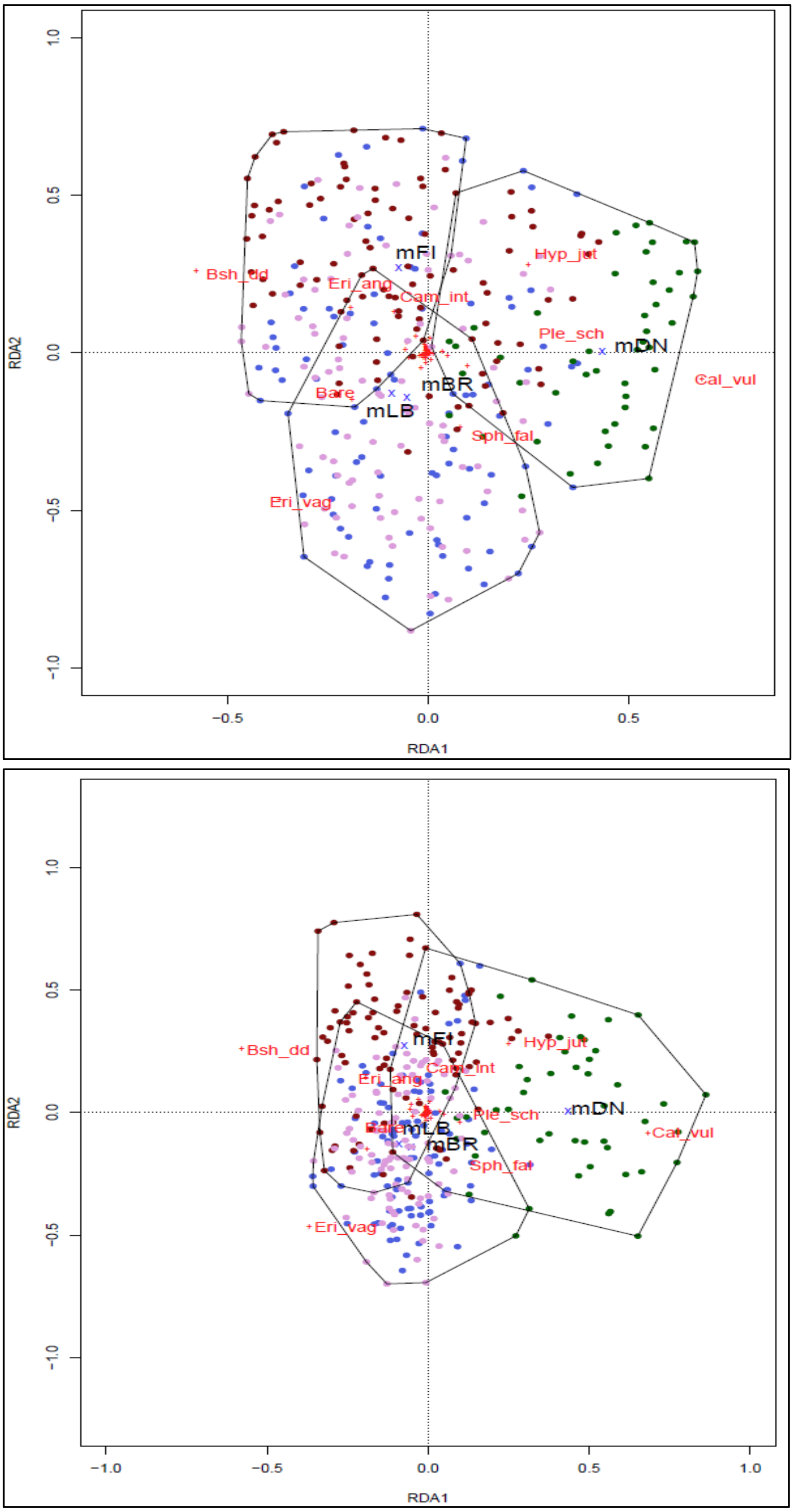


Fig. 41 Redundancy analysis (RDA) of vegetation composition in 2013 to 2016 (i.e. post-management) with management as the constraining variable (**top**) and including year, site and block added as random effects variables (**bottom**). Blue crosses represent the centroids of the management (m) groups with DN (uncut), FI (burnt), BR (mown, brash removed) and LB (mown, left brash). Red crosses represent the positions of individual species of which the eight most influential are labelled with the first three letters of the genus name and species name (for plant species; Bsh_dd is brash/dead/burnt material and Bare is bare ground). Coloured dots represent the position of each individual plot in a given year where dark green is for DN, brown is for FI, blue is for BR and pink is for LB. The groups assigned by k-means partitioning are shown by the polygons.

The RDA exploring *Sphagnum* species composition with management as the constraining variable and including year, site and block as random factors (**Figure 42**) explained only 5.9% of the variance of the *Sphagnum* species matrix (adjusted R^2 = 0.06, $F_{3,\ 395}$ = 12.08, p< 0.001). The first axis explained only a small proportion of the variability (7.6%) and the RDA did not show any meaningful groupings in respect to actual plot management effects on *Sphagnum* species composition.

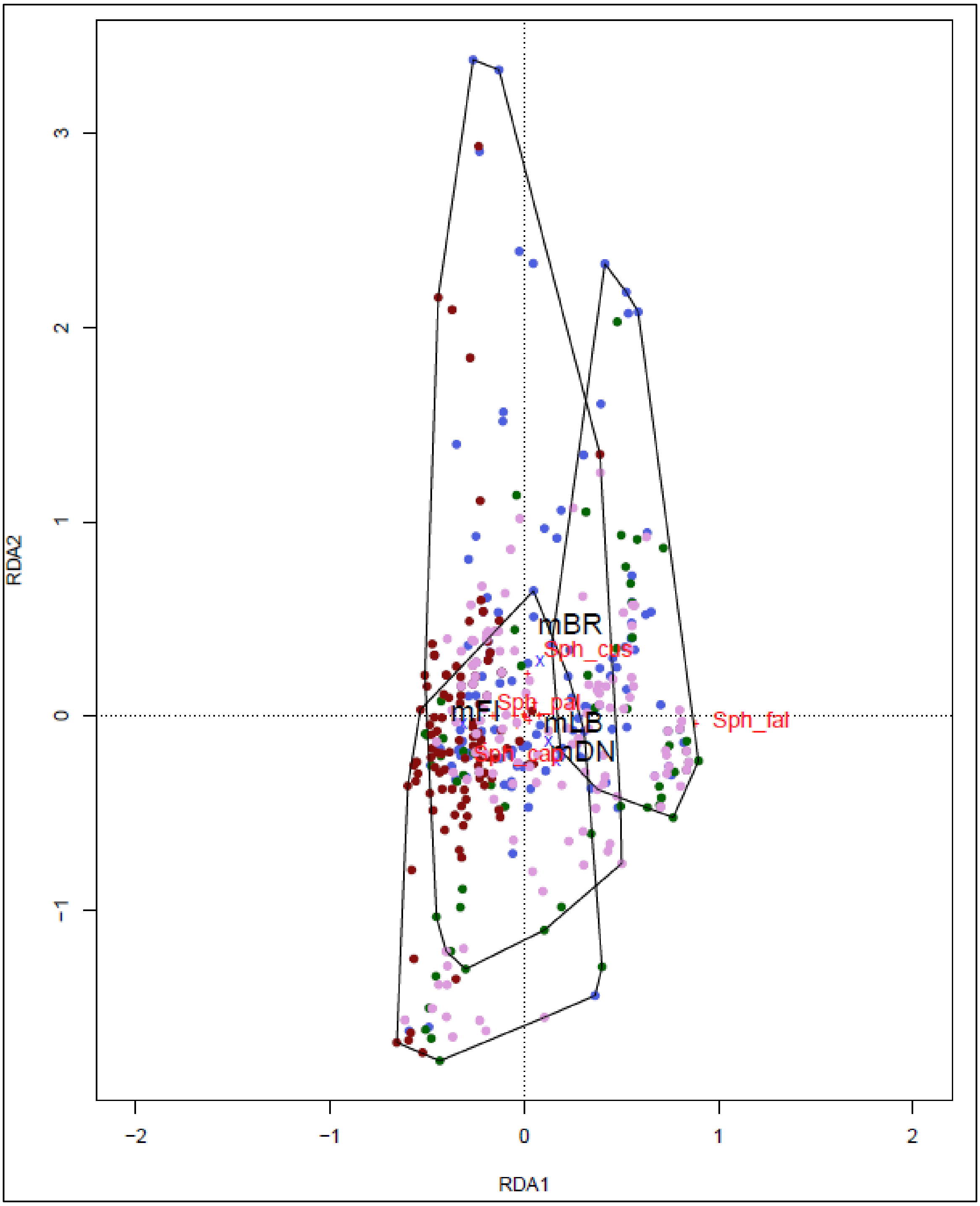


Fig. 42 Redundancy analysis (RDA) of *Sphagnum* composition in 2012 to 2016 (i.e. pre- and post-management) with management as the constraining variable, including year, site and block as partialled out conditions (i.e. used like random factors). Blue crosses represent the centroids of the management (m) groups with DN (uncut), FI (burnt), BR (mown brash removed) and LB (mown left brash). Red crosses represent the positions of individual *Sphagnum* species of which the four most influential are labelled with the first three letters of the genus and species name. Coloured dots represent the position of each individual plot in a given year where dark green is for DN, brown is for FI, blue is for BR and pink is for LB. The groups assigned by k-means partitioning are shown by the polygons, which appear not to relate to the plot management groups.

Discussion

These vegetation analyses confirmed that all the sites supported a blanket bog vegetation community and that the complement of species present at each site was similar, although Nidderdale showed lowest species richness. Notably, only a few plant species showed significant management related impacts (i.e. *Sphagnum* spp., *Eriophorum vaginatum* and *Hypnum jutlandicum*), although by chance some differences were already apparent pre-management, whilst other cover species/groups showed likely transient differences and no significant change in the number of species in relation to management (**Figure 40**). The high *Calluna* cover observed at all sites pre-management (**Figures 34** and **37**) is probably a result of the historic burning, evident at a site nearby Mossdale in the large and sustained increase in charcoal fragments over the last 200 years (McCarroll et al., 2015) and also confirmed for the other sites (see Section 4.3.3). Nevertheless, the differences in species richness and diversity between sites indicate that the sites were not as similar as they first appeared. Mossdale had the highest diversity and richness and visually appeared the least heavily managed, whilst Nidderdale had the lowest richness and diversity and appeared the most intensively managed. However, Mossdale was also the wettest site (MAP of 2030 mm) and Nidderdale the driest site (MAP of 1590 mm), meaning that climate may play a role in regulating species richness and diversity on blanket bogs. Moreover, burnt plots had significantly higher diversity (i.e. effective number of species; **Figure 40**) during post- than pre-management (albeit differences were small and diversity remained slightly lower post-treatment in FI plots at 1 x 1 m scale than in the other treatments), suggesting that burning increased species diversity in the short-term more than any other management (**Figure 40**). This finding is supported by Harris et al. (2011) who concluded that prescribed burning maintained diversity, although the study was for a species poor, 'dry' Peak District bog. There was a slight (but not significant) increase in species richness from pre- to post-management for all managements apart from the 1 x 1 m and 5 x 5 m uncut plots (**Figure 39**), suggesting that the removal of the dominant *Calluna* resulted in more light and space for understorey plants to grow (Grace & Woolhouse, 1973). However, to determine the long-term trajectories of the vegetation community, a longer period of monitoring is clearly needed, as is suggested by the changes in vegetation cover between 2013 and 2016 (**Figure 34**). This long-term monitoring need also relates to capturing changes in the vegetation cover rather than diversity, and in particular for key species/groups, that are more important for the characteristic transition toward an unmodified bog as shown in the additional ecological vegetation composition analysis provided in Appendix 3a. This finding agrees with Hancock et al. (2018) highlighting the need to monitor longer than 6 years to capture community changes on blanket bog in relation to management change.

Calluna is managed on moorlands in order to encourage new growth for red grouse to eat and burning has been advocated as the most effective method for at least a century (Lovat, 1911). In many areas this has led to an over-dominance of heather. Our study showed faster immediate regrowth of *Calluna* on mown than on burnt plots, recovery rates converged after two growing seasons (**Figure 34**) corresponding to delayed regrowth from seeds on burnt plots compared to stem regrowth on mown plots. This agrees with findings by Liepert et al. (1993) who found no difference in *Calluna* cover between mowing and burning. Moreover, Kayll and Gimingham (1965) assessed vegetative regrowth (i.e. production of shoots from existing stems) of *Calluna* plants two months after burning and mowing and found greater regeneration from cut plants which agrees with the present study. Further study over a full management cycle (~10-15 years) is required to determine whether mowing at particular times of year (or repeated mowing, as was anticipated in the experimental design) benefit species other than *Calluna*, and to assess whether regeneration on burnt plots is likely to outperform *Calluna* growth on mown plots in the future. Moreover, an assessment of potentially less re-growth after mowing older heather stands, as indicated by Hobbs & Gimingham (1984), was not considered. One advantage of mowing over burning seems to be a much smaller area of bare ground post-management (**Figure 34a,b**), particularly when leaving the brash (**Figure 35**), which potentially could reduce erosion and particulate organic carbon (POC) concentrations in streams (Evans et al., 2006; 2014).

The two cotton-grass species *Eriophorum vaginatum* and *E. angustifolium* are typical bog species, although their benefits for a bog are not as clear as those of *Sphagnum* mosses. On the one hand, *Eriophorum* species can take up substantial amounts of carbon (Bortoluzzi et al., 2006), can contribute to peat formation (Crowe et al., 2008; Bain et al., 2011), and rapidly colonise bare peat after disturbance (Crowe et al., 2008), thereby reducing runoff and increasing water retention time in the peat (Grayson et al., 2010). On the other hand, *E. vaginatum* can form dense tussocks, which may block out light beneath, and has aerenchyma which facilitate CH_4 release (e.g. Cooper et al., 2014) as well as aerating the peat, which may accelerate decomposition (Freeman et al., 2001). In this study, the majority of plots contained *E. vaginatum* both before and after management implementation, with some plots containing *E. angustifolium* at a much lower cover. Whitendale had more *Eriophorum* cover on uncut plots than either of the other sites (**Figure A3.2a-c** in Appendix 3), probably largely due to this site having least *Calluna* both pre- and post-management, which could reflect a shorter past burn rotation. *Eriophorum* cover increased on both burnt and mown plots following management (**Figure 34a,b**), but the much greater increase on mown plots strongly suggests that mowing encouraged it, *E. vaginatum* in particular. Previous studies (Hobbs, 1984; Lee et al., 2013) have found that *E. vaginatum* dominated when burning was on a short (10 year) rotation. Although it appears that no studies have specifically investigated the effects of mowing on *Eriophorum* species, Milligan et al. (2004) demonstrated that cutting reduced cover of *Molinia caerulea*, a moorland grass which often grows in tussocks. Therefore, it was unexpected that mown plots would have higher *E. vaginatum* cover than burnt plots, especially since it was observed during mowing that the blades removed the tops of many existing *E. vaginatum* tussocks (which might encourage more basal regrowth because of loss of apical dominance and hence increasing cover). However, *E. vaginatum* cover did also increase on burnt plots following management (**Figure 35B**; **Table 8**), but whereas this was similar to mown plots for the overstorey assessment (and was the case already pre-management), the total cover revealed a greater pre- versus post-management increase on mown plots.

In contrast to the similarities in species richness and diversity (**Figures 39** and **40**), the species composition changed over time under management intervention. The RDA, combined with the k-means polygons, showed that the burnt, mown and uncut groups each had a major and minor factor separating them, with another two factors in the overlap between the burnt and mown groups (**Figure 41**). Given that RDAs usually use many variables to attempt to explain changes or differences in vegetation cover (e.g. Field et al., 2014; González & Rochefort, 2014), it is remarkable that management alone, especially when combined across sites and years, explained 15% of the variability of the vegetation composition. Additionally, this was greater than the variability explained by site (13.4%), year (13.2%) or block (3.8%). This strongly suggests that management was instrumental in driving changes in species composition. Considering that a burnt and a mown polygon were clearly identified, these changes were not solely due to a reduction in *Calluna*. Importantly, these management related differences indicate possible different long-term trajectories in vegetation composition between burnt and mown areas, though longer term monitoring is needed to assess this. Other factors such as climate, atmospheric N deposition and landscape features (e.g. slope, aspect, etc.) may also have influenced the vegetation cover but are far more likely to have acted across all plots on a site, regardless of management. However, it is not necessarily the number or the evenness of species which ultimately matters for a bog, it is which key species are affected that matters most. As peat represents a vast and stable carbon store (Gorham, 1991) and is largely composed of and maintained by *Sphagnum* moss (Bain et al., 2011), *Sphagnum* is clearly such a key species group on blanket bog.

The project's main aim was to assess management options to reduce heather cover and encourage 'active' bog vegetation, specifically *Sphagnum* spp. Of particular interest in this respect is the observed increase in *Sphagnum* spp. total cover in the final year (**Figure 34b**). It is generally thought that burning retards *Sphagnum* growth, cover and carbon uptake (Kuhry, 1994; Campeau & Rochefort, 1996). Whilst a negative impact of burning on total *Sphagnum* cover was not found, the total cover assessment indicated an increase in *Sphagnum* spp. cover on mown plots, especially in the last year (**Figure 34b and Figure 37**). The relative scarcity of *Sphagnum* at the two

drier sites (see **Figure A3.2a-c** in Appendix 3) may be important as lower water tables (i.e. drier) and peat moisture may be an impediment, or delaying factor, in the development of a trajectory towards more 'active' blanket bog with high *Sphagnum* cover, likely resulting in *Calluna* regaining dominance too quickly unless there is a relatively rapid post-treatment 'natural' increase of *Sphagnum* or the re-introduction of *Sphagnum* propagules. Thus, if mowing or burning is used as a restoration treatment when *Sphagnum* is absent or very scarce then it may need to be added. Although this aspect was included in the project's experimental design (i.e. BeadaMoss pellets), it clearly requires a longer period to establish and continuation of the vegetation monitoring. *S. fallax*, which was the only *Sphagnum* species showing influence in the RDA (**Figure 41**), had the lower cover on burnt plots post-management; however, burnt plots also had lowest cover of this species pre-management and there was a small increase in *S. fallax* cover after burning. Additionally, when the cover of all *Sphagnum* species (excluding *S. fallax*) was combined, *Sphagnum* cover was relatively constant across periods without any meaningful management influence on *Sphagnum* spp. composition (**Figure 42**). However, by chance, burnt plots had less *Sphagnum* cover than the other managements before management implementation (**Figure 35**) but there was an indication that this cover was maintained even three years after burning took place. Clearly, longer term monitoring is required to fully assess the effects of burning on *Sphagnum* cover and to determine whether the brash on LB plots enhanced *Sphagnum* growth by protecting *Sphagnum* fragments and plants from desiccation. Moreover, the higher proportion of other moss species, particularly *Hypnum* and *Campylopus introflexus*, on burnt plots is ecologically important. Together with higher brash/dead/burnt material cover on burnt compared to mown plots this indicates that burning maintained the current habitat status whilst mowing seemed to initiate a different trajectory towards 'active' bog vegetation (i.e. greater *Eriophorum* and *Sphagnum* and less non-*Sphagnum* moss cover) although so far without any impact on heather cover or species richness.

In summary, the vegetation assessment described in Sections 4.2.5 and 4.2.6 showed that:

- The nutritional value of heather shoots was improved equally by mowing or burning, compared to uncut plots, although comparison to potential long-term rejuvenated, layered heather regrowth remains unknown.
- Whilst for most nutrients levels in all treatments were sufficient to satisfy the requirements of grouse, for phosphorus and manganese, active management, for example, by either burning or mowing may be needed to meet the needs of red grouse.
- *Sphagnum* pellet introduction did result in only a very limited visual establishment during the project period, which could reflect slow establishment from small fragments (inside the pellets) and detectable growth under upland field conditions likely requiring more than 5 years.
- Heather re-growth after management was initially slower on burnt than mown plots. However, 4 years after management intervention, both cover and height of heather were similar on burnt and mown plots.
- Over 4 years since management intervention, the species composition of both burnt and mown plots has overall moved closer towards that of the uncut plots, but there was an indication of possible different long-term trajectories (with mown plots possibly becoming a more 'active' bog).
- However, it is unclear at this time whether the vegetation cover and composition of burnt and mown plots found after 4 years will remain as the possible trajectory towards vegetation maturity continues.
- The cover of *Eriophorum vaginatum* increased on both burnt and mown plots after management intervention. Its cover was greater on mown than burnt plots after management intervention, but this was also the case before management though less so for total cover.
- The combined area of bare/brash/burnt ground was significantly greater on burnt than mown plots during the first two years after management.
- Four years after intervention, there was little difference between management treatments in the cover of bare, burnt or brash-covered ground.

- The total cover of *Sphagnum* species across all sites was not significantly affected by the management intervention after 4 years, although there was an initial increase in exposed 'top view' cover on the mown plots and a strong indication of higher total *Sphagnum* cover on mown plots 4 years after management at the two wetter sites, particularly at Mossdale. Management intervention also did not affect *Sphagnum* species composition.
- The cover of non-*Sphagnum* mosses, which were dominated by *Hypnum jutlandicum*, was higher on the burnt than the mown plots after management, but this was also the case before management and it increased most on the uncut plots. Although *H. jutlandicum* cover was similar throughout, *Campylopus introflexus* cover increased post management, especially on burnt plots, though cover was fairly low.
- Whilst the driest and most modified site, Nidderdale, had the lowest overall *Sphagnum* cover, it had the highest number of *Sphagnum* species, likely reflecting greater habitat variability in wetness. The wettest, least modified site, Mossdale, had by far the greatest cover of *Sphagnum*.
- The long-term aim of whether the treatments were successful in reducing heather cover and encourage 'active' bog vegetation can only be assessed over a longer monitoring period (addressing slow re-growth and transient effects). Moreover, repeated mowing (<10 years) could shift the balance towards a less heather dominated vegetation quicker than a one off mowing as part of a usual management rotation (>10 years), which could be assessed by introducing repeat mowing and continued monitoring.
- In addition, a possible assessment of cutting older heather to encourage potentially less vigorous re-growth from dormant buds could be considered alongside repeated mowing as a potential management to reduce heather cover.

Moreover, the more in-depth ecological assessment on species cover and changes over time in relation to site, sub-catchment and management (provided by R.A. Lindsay) as presented in Appendix 3a can be summarised as follows:

- The three sites differ from each other, with Nidderdale in the driest, most degraded condition, Whitendale supporting a vegetation which is most uniformly of a 'typical bog' community though still dry and somewhat degraded, while Mossdale is the wettest site, though this mainly arises from the presence of poor-fen vegetation characterised by *Sphagnum fallax* and most likely associated with recovering (micro-) erosion features or possibly poor-fen flush systems.

- Within each study site, the plots selected for the burning treatment consistently differ in their initial condition from those subject to other treatments, being generally more species-poor and supporting fewer typical bog species. This distinction is particularly marked in the 1 x 1 m quadrats but is also true of the 5 x 5 m quadrats.

- Over time, the invasive moss *Campylopus introflexus* became established and often then expanded on the burning treatment plots of all three sites, particularly at Nidderdale. This moss is particularly associated with dry bare peat. Nidderdale and Mossdale also showed an increased presence on the burning plots of grass and herb species such as *Anthoxanthum odoratum*, *Deschampsia flexuosa*, *Galium saxatile* and *Festuca ovina*, which generally point to heavily modified bog conditions – indeed more grass heath than blanket bog. Whitendale does not show this trend so markedly, but instead shows a marked increase in

leafy liverworts which are generally associated with drying conditions. In general, the burnt quadrats had less overall cover of typical ‘bog’ *Sphagnum* species than the other treatments.

- Background trends as recorded in the larger uncut ‘no treatment’ 5 x 5 m quadrats indicate that *Hypnum jutlandicum* has increased substantially at Nidderdale and somewhat at Mossdale, though without an equivalent increase in cover of *Calluna vulgaris*. Such a trend suggests that the bryophyte layer may be re-establishing itself on drier parts at these sites, albeit not yet as a peat-forming vegetation. At Mossdale there is also an increase in *Sphagnum fallax*, suggesting similar recovery within micro-erosion gullies but in this case with a peat-forming sward. Whitendale, meanwhile, shows a small but steady increase in *Sphagnum papillosum* and *Eriophorum angustifolium*, which points to slow recovery of ‘typical’ bog vegetation.

- The most striking effect of both mowing treatments at Nidderdale and Whitendale is the increased cover of *Eriophorum vaginatum* and the increased presence of poor-fen species, the former as a result of reduced competition from *Calluna vulgaris* while the latter may have taken advantage of enrichment from brash cover or fragments of brash left after mowing. At Mossdale there is increased cover of *Sphagnum fallax*, suggesting enhanced recovery of micro-erosion gullies and/or possible enrichment from brash cover or fragments, together with a marked increase in *Eriophorum vaginatum* and *Sphagnum capillifolium*, both of which may have benefitted from reduced competition with *Calluna vulgaris*.

- In summary, therefore, burning appears to be the least beneficial form of management intervention. The uncut ‘do nothing’ option has few, if any, downsides although on the most damaged (i.e. more modified) site (Nidderdale) there is only limited evidence for recovery of a bryophyte layer which, in due course, can be expected to give way to a peat-forming bryophyte layer. Mowing encourages expansion of hare’s-tail cotton grass (*Eriophorum vaginatum*) which is a key species in re-establishment of a peat-forming community, albeit on rather long timescales. Where such expansion is also accompanied by an increase in *Sphagnum capillifolium* the timescales of recovery to a more ’typical’ bog vegetation may be significantly reduced, but this is only likely on wetter sites – such as Mossdale. Whether brash is left on-site or not does not appear to have a substantial effect on the vegetation response.

4.2.7 *Hydrological conditions*

The installed water table depth meters (WT-HR 1000, TruTrack, New Zealand) inside the dipwells (**Figure 43**) were checked regularly, and water table depths (WTD) were corrected, based on manual offset measurements using a fibre optic rod lowered inside the dipwells (see **Figure 43**).

Fig. 43 Dipwell (**left**) within a wire mesh covered area containing an Omnilog water table sensor, a pore water Rhizon sampler (covered by the green mesh) and a temperature sensor (covered by the reflective shield). Also shown is a syringe used for sampling pore water from the Rhizon sampler. A manual fibre optic rod (**right**) was used to measure water table depths in the dipwells by conveying light to the water level within the dipwell (i.e. the light reflection stopped as soon as the water fibre optic rod entered the water level).

Overall monthly average WTD for uncut (DN) plots (**Figure 44**) showed marked seasonal variation, with the wettest periods generally occurring during January and February and the driest periods between June and September. Moreover, there were clear site differences in WTD on the DN plots, with Mossdale being the wettest overall and Nidderdale being significantly drier than the other two sites ($p < 0.005$).

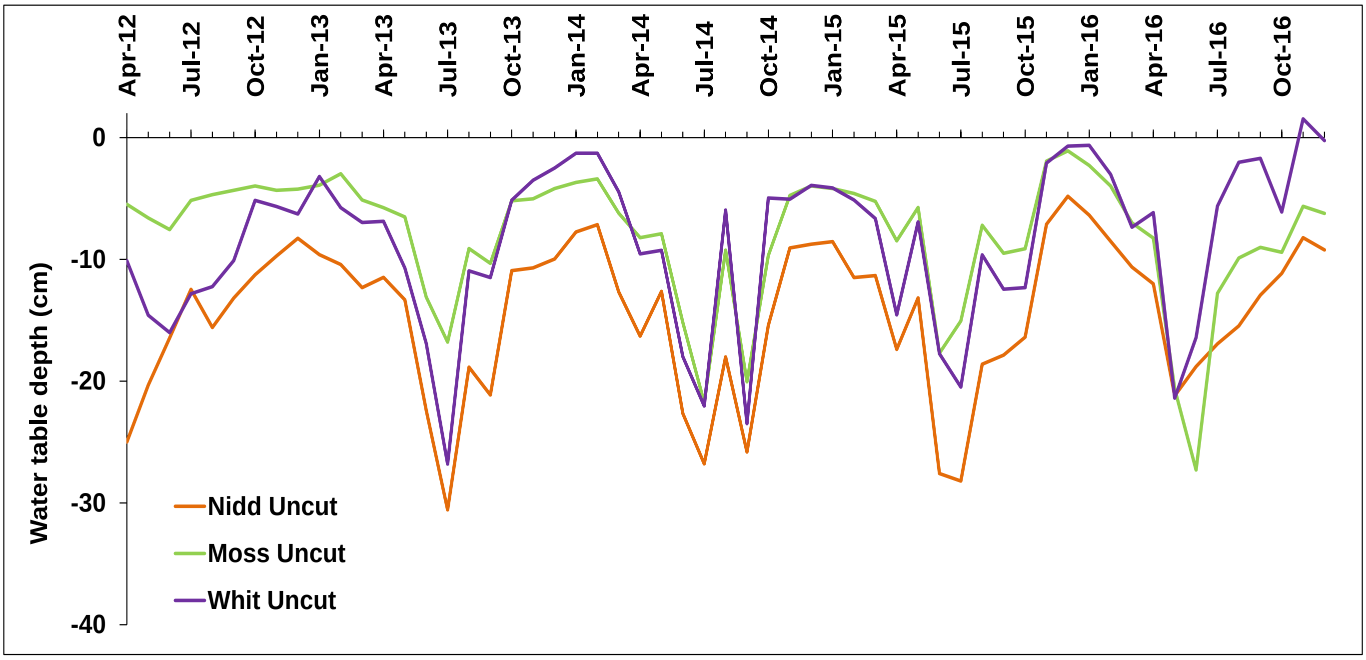


Fig. 44 Average monthly water table depths (WTD) measured at the uncut (DN) plots at Nidderdale (Nidd), Mossdale (Moss) and Whitendale (Whit) during 2012-2016. Average WTD ± standard deviation (STDEV) for the uncut plots over the five years was: Moss -8.1 ± 5.7 cm; Whit -8.7 ± 6.9 cm; Nidd -14.6 ± 6.4cm.

However, although uncut plots at Mossdale were the wettest during the 2012-2013 pre-management period (-7.3 cm) and the driest at Nidderdale (-13.2 cm), uncut plots at Whitendale became wetter from 2013 onwards and were the wettest during 2016, resulting in a very similar 2013-2016 post-management average monthly WTD of -8.6 cm compared to -8.0 cm for the uncut plots at Mossdale (see **Table 9**). However, all three sites showed a general reduction (drier) in WTD across all managements between pre- and post-periods (**Table 9**). The statistical output in this project period related to the main question if management alters the overall WTD. Therefore, we divided pre- versus a combined post-management period. However, with a longer monitoring period splitting the post-management period into separate phases (e.g. in sections of 3 years) would become feasible, but so far only four post-management years were measured. There were significant differences (reported F-ratios and p-values) in the average daily WTD between managements (with the +/- *Sphagnum* treatments combined) in the two periods ($F_{3,\ 123721}$ = 28.5, $p < 0.001$) with LB having the highest post-management WTD (-8.9 cm), followed by BR (-10.3 cm) and DN (-10.5 cm), with FI plots having the lowest post-management WTD (-13.2 cm; which was reduced to -10.8 cm if one burn plot (No. 4, see **Table 4**) at Whitendale with a very large peat pipe and overall very low WTD was removed). This corresponded to about 3.0 cm lower average post-management WTD on burnt compared to mown (LB) plots (average of 1.7 cm and 4.2 cm depending on removing or including the Whitendale FI plot with very low WTD, respectively). However, the post-management difference between burnt and mown (LB) plots for only Nidderdale and Mossdale was 3.3 cm. This indicated less evaporation from mown areas with the brash left (LB) than from uncut areas (DN) and mown areas with the brash removed (BR), as well as increased runoff from burnt (FI) areas. Similarly, there was a significant interaction between management, site and time period ($F_{6,\ 123721}$ = 145.2, $p < 0.001$), although managements did not appear to cause consistent changes in WTD at all sites. For example, the positive brash effect (i.e. wetter if leaving brash) was mainly observed at the driest site (Nidderdale). Moreover, (for Whitendale) some significant differences were also observed pre-management, suggesting that the differences were not entirely caused by the management. However, these findings agree with results based on a single site in England (Worrall et al., 2013), indicating highest WTD for mown with brash left and lowest under burning or previously burnt patches. Also, as cited in Glaves et al. (2013), White et al. (2004) and Yallop et al. (2008) reported lower WTD under *Calluna* and lower and more oscillating WTD under recent burns, as did the EMBER project (Brown et al., 2014) for recent burns with recovery by >10 years post burn.

Table 9 Mean ± one standard deviation (STDEV) of monthly water table depth (WTD) by site and by management during the pre- (2012-2013) and post- (2013-2016) management period. Managements were uncut (DN), burnt (FI) and mown with brash left (LB) or with brash removed (BR), including the +SP (*Sphagnum* pellet addition) treatments.

		Nidderdale				Mossdale				Whitendale		
	DN	FI	LB	BR	DN	FI	LB	BR	DN	FI	LB	BR
Pre-management	**-13.2**	**-12.6**	**-10.4**	**-12.4**	**-7.3**	**-7.7**	**-5.8**	**-5.5**	**-9.8**	**-14.0**	**-8.5**	**-10.6**
(2012-2013) STDEV	4.9	4.6	4.5	4.4	1.2	3.0	3.3	2.6	4.2	5.1	6.2	4.2
Post-management	**-14.8**	**-13.3**	**-10.1**	**-13.7**	**-8.0**	**-10.5**	**-7.1**	**-8.2**	**-8.6**	**-15.8**	**-9.7**	**-9.0**
(2013-2016) STDEV	6.5	9.6	6.1	6.5	5.8	6.9	6.2	5.7	7.1	6.0	7.3	7.4

The plots of monthly mean WTDs shown in **Figure 45** show the response to management intervention. Mossdale showed a reduction in WTD on the burnt (FI) plots during summer 2013 immediately after management, whereas this was only observed at Nidderdale during the subsequent years (2014 and 2015). Moreover, whereas Nidderdale showed a wide range in WTD across treatments, Whitendale showed remarkable narrow ranges (apart from the burnt plots). However, when removing the Whitendale FI plot with very low WTD throughout the study period (plot No. 4; see comment above), burnt plots pre- and post-management (-10.7 cm and -8.2 cm, respectively) had very similar WTD compared to other managements.

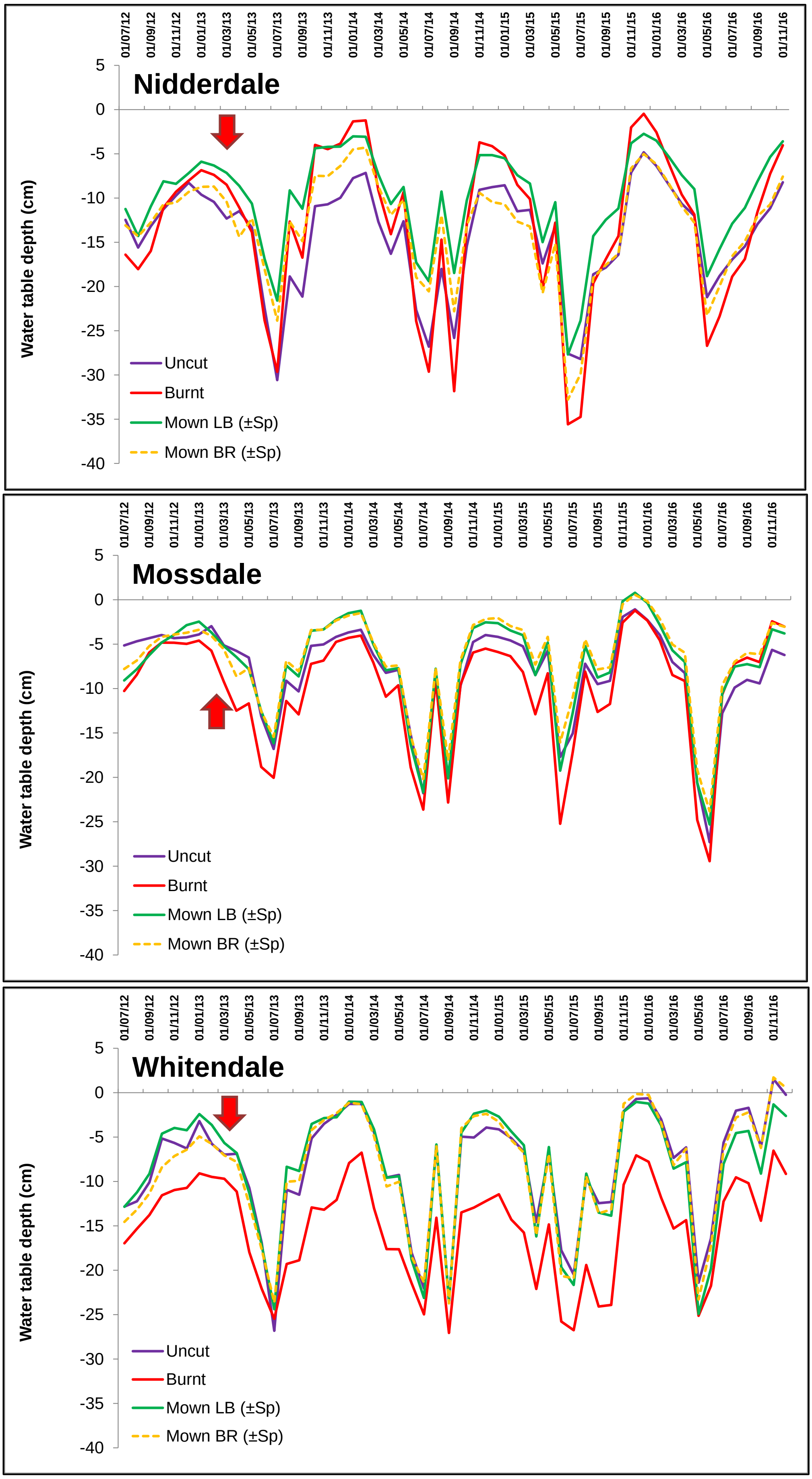


Fig. 45 Average monthly water table depths (WTD) measured at Nidderdale (**top**), Mossdale (**middle**) and Whitendale (**bottom**) during 2012-2016 at the management plots. Managements were Uncut (DN), Burnt (FI), Mown either with brash left (LB) or brash removed (BR) for the combined *Sphagnum* pellet (+Sp) addition. The red arrow marks the onset of the burning and mowing management in March 2013.

The average monthly WTDs were significantly impacted by month of measurement ($F_{11,\ 3605}$ = 249, p< 0.001) and there was a highly significant interaction between management, site and time period (see median values and ranges in **Figure 46**; $F_{6,\ 4067}$ = 6.44, p< 0.001). Again, the post-management period was analysed as combined data.

When monthly WTD data was tested separately for each site (see **Figure 46**), there were still significant interactions between management and time periods but Whitendale had the most significant interaction ($F_{3,\ 1347}$ = 8.23, p< 0.001), Mossdale the next significant ($F_{3,\ 1349}$ = 4.18, p = 0.006) and Nidderdale the least ($F_{3,\ 1349}$ = 2.87, p = 0.035). There were no significant differences between managements pre-management, apart from at Whitendale where LB (mown with brash left) plots were significantly wetter than FI (burnt) plots (p = 0.01). Post-management, this difference was still present, but LB plots were wetter than FI plots at Mossdale and Nidderdale as well (p< 0.001 for both). Moreover, at Mossdale and Whitendale, FI plots also had a significantly lower WTD than BR (mown, brash removed) and DN (uncut) plots (p< 0.02 for all), whereas at Nidderdale, FI plots were slightly, but not significantly, wetter than BR and DN plots. Additionally, LB plots had significantly higher WTD than BR or DN plots at the dry site Nidderdale (p< 0.001 for both) but were only marginally higher than BR plots at the very wet site Mossdale (p< 0.070), suggesting that leaving the brash reduced evaporation rates, particularly at the dry site. Although this brash effect cannot be assessed in detail with the current analysis, the brash layer could still be observed in 2016, compacting and re-vegetating (mostly with bryophytes and sedges) since management. However, at Whitendale LB plots were slightly, but significantly drier than DN plots (p< 0.001) and slightly drier than BR plots. With an average annual WTD of around -7 cm, Mossdale seemed to be the most "intact" or least modified blanket bog site as this value was very similar to the average annual WTD at the unmanaged Moor House site (Heinemeyer et al., 2010; Heinemeyer & Swindels, 2018). The overall range of WTD at Mossdale (see **Figures 46**) was very similar to that reported by Holden et al. (2015) with similar seasonal patterns and a summer low (see **Figure 45**) generally observed on blanket bogs. Daily minima at Moor House are observed to -40 cm during particularly dry summers (Evans et al., 1999) and can be even lower on degraded drained sites (Wilson et al., 2010).

As visually the WTD drawdown seemed to occur mostly during summer, daily WTD differences were also compared separately for summer (June to August) and winter (November to January) periods. There were few clear seasonal differences, as all managements differed significantly (p< 0.001) from each other in both summer and winter during the post-management period, apart from between FI and DN plots in summer at Nidderdale, DN and BR plots in summer at Mossdale and LB and BR plots in winter at Whitendale. However, whilst during post-management summer WTDs were lowest for burnt plots on all sites (see **Figure 45**; WTD on burnt plots was 3.4 cm for Nidderdale, 2.4 cm for Mossdale and 7.0 cm for Whitendale lower than the average WTD across all other plots), during winter this was similar with 1.6 cm at Mossdale and 8.4 cm for Whitendale, but Nidderdale burnt plots were on average 3.2 cm wetter.

Although there are reported values of restoration impacts on WTD recovery by drain blocking (Wilson et al., 2010) and revegetation (Dixon et al., 2014), there is only one other study on the WTD impacts comparing uncut, mown and burnt treatments (Worrall et al., 2013). However, whereas the latter study also observed a reduction in WTD on burnt plots compared to mown plots, we did not observe a much lower WTD for the uncut plots as reported by Worrall et al. (2013), either for monthly (**Figure 46**) nor for daily (**Figure 47**) WTD.

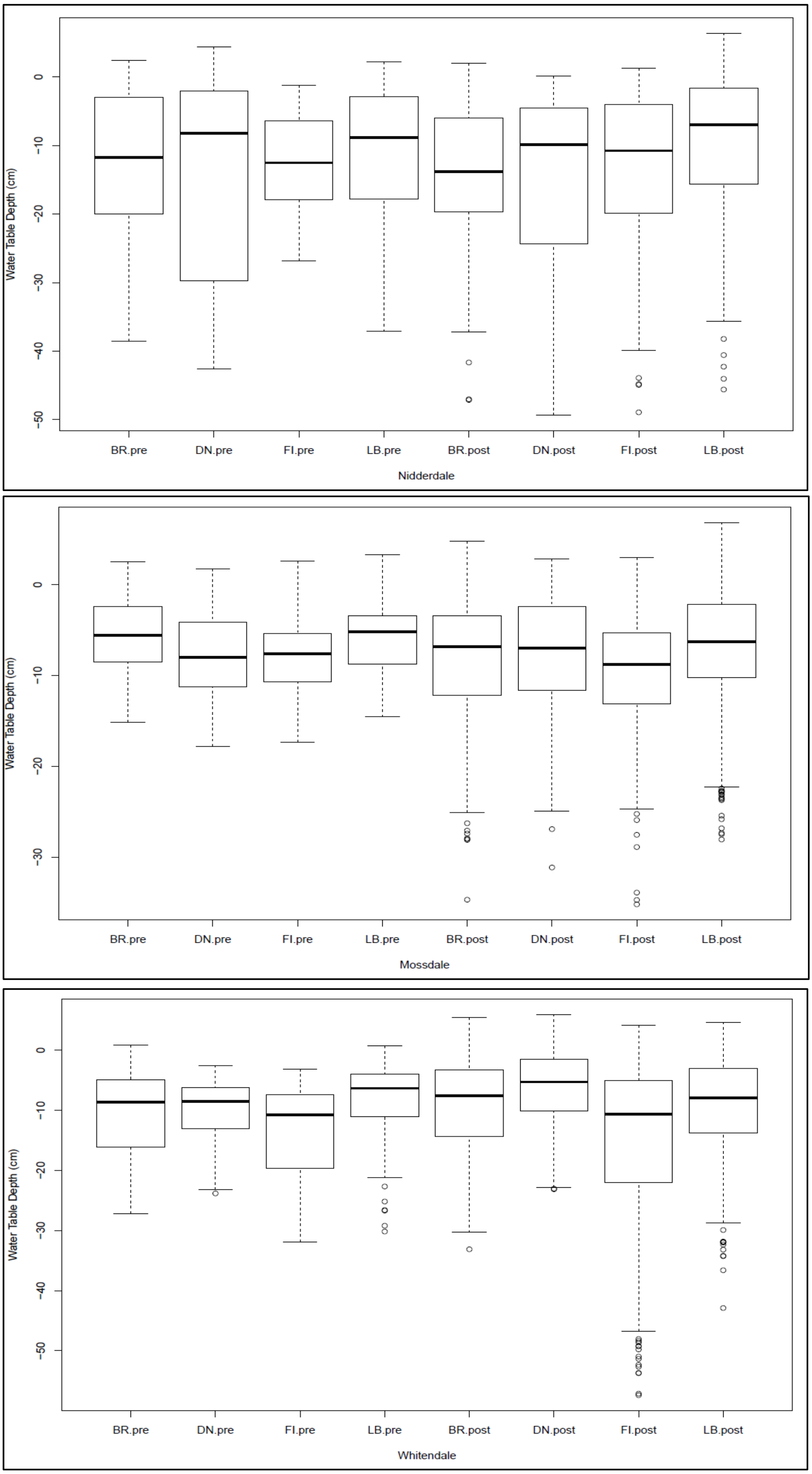


Fig. 46 Monthly water table depth (cm) ranges for the main managements (i.e. with ±*Sphagnum* addition combined) for each site (**top**: Nidderdale; **middle**: Mossdale; **bottom**: Whitendale) during the pre- (2012) and post-management (2013-2016) periods. The box midline indicates the median, the box edges indicate the interquartile range and the points indicate data less than 1.5 times the interquartile range. Managements were Burnt (FI), Uncut (DN), Mown (LB, brash left) and Mown (BR, brash removed).

The daily WTD data shown in **Figure 47** indicate that Mossdale had the wettest and most stable conditions, with Nidderdale and Whitendale having drier conditions, with lower values of WTD being reached. The lowest daily WTD was about -60 cm at Mossdale, -70 cm at Nidderdale and -80 cm at Whitendale. Comparisons of post-management suggested that the WTD interquartile ranges were overall slightly smaller for the uncut (DN) and mown (LB & BR) areas compared to those for burnt plots (**Figure 47**; bottom), indicating potential enhanced drying and water loss via runoff by burning. Moreover, the post-management monthly (**Figure 46**) and daily median WTD values and interquartile ranges (**Figure 47**; bottom) were always higher (wetter) for the LB than BR plots, indicating that the median captured the brash effect better than the monthly averages (cf. **Figure 45**).

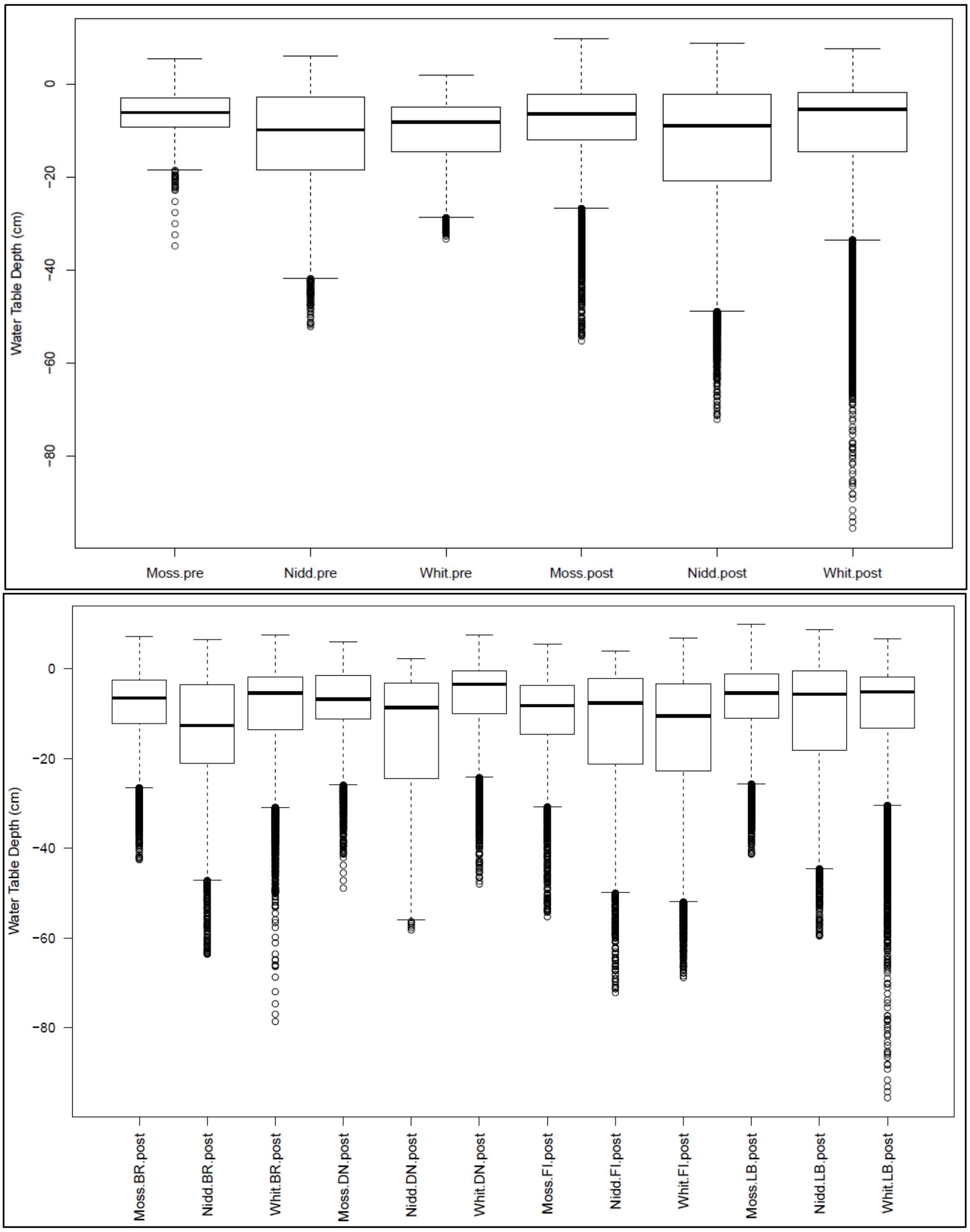


Fig. 47 Daily water table depth (cm) ranges, averaged across the three sites (Mossdale, Moss; Nidderdale, Nidd; Whitendale, Whit), during the pre- (2012) and post-management (2013-2016) periods (**top**) and for the post-management period for each site and management (**bottom**). The box midline indicates the median, the box edges indicate the interquartile range and the points indicate data less than 1.5 times the interquartile range. Managements were Burnt (FI), Uncut (DN), Mown (LB, brash left) and Mown (BR, brash removed) with ±*Sphagnum* treatments combined for the mown plots.

4.2.8 *Stream flow*

The same Omnilog sensors that were used for the plot WTD measurement (see Section 4.2.7) were used to measure flow rates at V-notch weirs (for pictures see **Figure A4.1** in Appendix 4) for each sub-catchment (see Appendix 4 for a detailed methodology). Manual measurements (see **Figure 48**) were used to correct for any measurement offsets. Two loggers were installed at each location to prevent any data loss due to logger failure.

Fig. 48 A V-notch flow weir being measured with a ruler to manually record the water height to allow the offset correction for the continuous logging of the water table by the Omnilog logger (within the white perforated PVC tube). A sample bottle contains the flow sample. For other pictures of all flow weirs, please see Figure A4.1 in Appendix 4.

The flow weirs provided continuous hourly flow rates during both the pre- and post-management period (**Table 10**). However, data for the first half of 2012 (until mid-July) were gap-filled using a regression between monthly rainfall and flow rates, as flow weirs were only installed in July 2012. Overall, 2013 was a very low flow (dry) year compared to 2014 which had a very high flow. This is in agreement with the rainfall totals, which showed a particularly wet first quarter in 2014, with the overall highest monthly flow rates occurring during the very high rainfall in December 2015 (see **Figure 10**). The pre-management period was characterised by remarkably similar flow rates between each pair of sub-catchments (**Table 10**). Across all years and sub-catchments, mean flow rates were lowest at Nidderdale (~14,300 m^3 ha^{-1} y^{-1}), higher at Mossdale (~15,900 m^3 ha^{-1} y^{-1}) and highest at Whitendale (~16,200 m^3 ha^{-1} y^{-1}). However, soon after the onset of different managements in April 2013, the burnt “control” (C) sub-catchments at Nidderdale and Mossdale showed higher flow volumes compared to those from the mown “treatment” (T) sub-catchments, although there was no noticeable difference at Whitendale. This may because at Whitendale both catchments received some mowing (on about equal areas) in 2009 (see extended mown strips in the aerial pictures shown in **Figure 2b**).

Using equations from the monthly version of the MILLENNIA model (Carroll et al., 2015; see Appendix 15) allowed calculating water volumes reflecting changes in WTD considering the available pore space in relation to the observed higher average monthly WTD on mown compared to burnt plots of around 3.3 cm (across Mossdale and Nidderdale only, excluding the Whitendale site with some previous mowing and generally less successful burns and a burnt plot of very low WTD), this translated to around 2.0 cm water (20 L m^{-2}); this monthly mean value equates to 200 m^3 ha^{-1} and over a year amounts to 2400 m^3 ha^{-1}. This annual amount of water potentially held within the peat on mown plots could be compared to the overall annual stream flow losses per catchment area on an equal per hectare basis (**Table 10**) after accounting for the increased proportion of sub-catchment area which received management intervention. After management, this annual amount of water was an average of 8% (2013), 4% (2014), 19% (2015) and 29% (2016) of flow volume per hectare from the mown catchments in Nidderdale and Mossdale; the increase in 2015/2016 reflects the second management intervention in early 2015.

Table 10 Annual (white rows) and monthly cumulative flow volumes for each sub-catchment at each site (C = Control-burnt; T = Treatment-mown). Flow volumes are presented in $m^3\ ha^{-1}$ taking into account the different sub-catchment sizes. Pre-management flow rates are highlighted in pale blue, post-management periods are highlighted in darker blue with mid-blue indicating flow rates after the first management application in spring 2013 and dark blue indicating the period after the second management application in spring 2015). Data for 2012 (January to July) were estimated based on a regression between monthly rainfall and flow rates during the pre-management period and available climate records (see Appendix 1).

Year/Month	Nidd C ($m^3\ ha^{-1}$)	Nidd T ($m^3\ ha^{-1}$)	Moss C ($m^3\ ha^{-1}$)	Moss T ($m^3\ ha^{-1}$)	Whit C ($m^3\ ha^{-1}$)	Whit T ($m^3\ ha^{-1}$)
2012	**14144**	**14507**	**15977**	**15812**	**16401**	**16105**
1	1388	1402	1756	1681	1616	1525
2	1055	1085	1235	1220	1197	1191
3	809	873	674	761	874	999
4	1131	1163	1137	1149	1137	1173
5	959	1010	969	1012	1010	1096
6	1370	1381	1634	1571	1581	1488
7	676	720	856	906	1069	1037
8	712	753	935	1038	901	745
9	1337	1343	1701	1671	1978	1866
10	1186	1234	1067	1110	1606	1602
11	1283	1336	1459	1393	1124	1109
12	2239	2206	2554	2301	2308	2274
2013	**7852**	**6890**	**11809**	**9328**	**8289**	**8770**
1	919	943	1243	1063	1105	1137
2	593	664	534	477	758	793
3	369	380	277	272	228	231
4	343	440	642	644	370	368
5	537	595	700	606	491	510
6	13	32	161	120	247	281
7	182	141	501	321	419	414
8	692	470	1083	664	751	872
9	200	152	487	342	561	572
10	1643	1195	2070	1555	1159	1251
11	691	519	1277	957	748	940
12	1673	1359	2835	2307	1451	1402
2014	**18597**	**14751**	**25661**	**19727**	**18541**	**19688**
1	3454	2751	4592	3734	3185	3344
2	4272	3657	5483	4448	3163	3154
3	1172	966	1479	1184	1239	1235
4	671	502	824	625	713	756
5	1765	1361	1539	1161	944	967
6	113	103	266	183	389	459
7	178	147	34	18	169	240
8	452	350	1342	980	1903	2271
9	66	41	38	16	243	315
10	1585	1170	2309	1705	1710	1828
11	1831	1359	2128	1489	1371	1360
12	3036	2344	5628	4187	3511	3759
2015	**11710**	**8970**	**19665**	**14801**	**13710**	**14310**
1	1490	1164	3062	2211	1788	1905
2	583	420	999	756	609	591
3	930	647	1242	994	862	860
4	196	142	591	427	303	304
5	693	544	1099	806	1018	971
6	189	150	266	199	263	379
7	169	118	386	304	258	252
8	493	381	1146	888	472	502
9	95	73	126	72	223	247
10	704	537	627	489	523	382
11	2715	2104	4005	3052	3004	3192
12	3453	2690	6117	4604	4387	4726
2016	**8803**	**5692**	**13234**	**10150**	**11461**	**12022**
1	2149	1605	2473	1886	1981	1949
2	1242	919	2382	1893	1967	1871
3	867	628	1092	873	985	932
4	476	330	697	538	492	478
5	75	58	52	29	185	190
6	469	346	101	60	589	565
7	340	256	656	474	805	977
8	659	328	1615	1260	1546	1774
9	462	202	1051	773	631	701
10	365	173	329	224	280	342
11	1144	569	1614	1268	1361	1545
12	556	278	1173	872	638	697

4.2.9 *Water balance*

The flow rates, together with the rainfall totals obtained from the weather stations (see **Figure 49**), were used to calculate the percentage of water lost via the streams for each sub-catchment. Calculations were only possible from July 2012, when the flow weirs were installed and assessed as change over time as management (and their impacts) increased over the entire sub-catchment. Water fluxes from chambers (**Figure 49**) were anticipated to provide evapotranspiration data to aid interpretation of the water balance calculations, but these data were not of high enough resolution (yet as vegetation regrowth was similar, evaporation should have been similar, too).

Fig. 49 From left to right: Weather stations were used to monitor incoming rainfall, water table depth was monitored using dipwells at the plot water sampling locations, flow weirs measured water outflow and chambers measured evapotranspiration. Measurements from these devices were intended to be combined to estimate the water balance.

The monthly percentage water losses (**Table 11**) were close to 100% during winter months in most years, suggesting the peat was saturated. In fact, at the start of each year, losses were sometimes greater than 100%, which can be explained by continued high water loss from a near saturated peat column during periods of low precipitation or could reflect a delayed snow melt response. During the summer months, values were more variable; although losses were usually about 10-30%, they sometimes fell to less than 10% during particularly dry summer periods, indicating high evapotranspiration rates and/or high soil water deficits. During spring and autumn periods, water losses were intermediate, between about 40% and 70%.

Importantly, calculated losses during the pre-management period (until April 2013) were very similar between the paired sub-catchments at all sites, providing ideal conditions for observing any management impact on flow. In particular, after the onset of burning and mowing in 2013, water losses were on average 10% higher in the burnt control (C) than in the mown treatment (T) sub-catchments at Nidderdale and Mossdale (but not at Whitendale), and this difference increased further to about 20% after the second management intervention in 2015 (**Table 12**). These values were consistent with the calculated annual flow losses over time of 8%, 4%, 19% and 29% (2013-2016, respectively) based on plot-level WTD changes (see end of section 4.2.8). Overall, this suggests that burning does reduce rain infiltration, as reported by Holden et al. (2013), and also that mowing can reduce this impact, holding back significant amounts of water.

Notably, there are hardly any such upland stream flow or water loss data available, meaning that these data provide an important insight into land management impacts on both water storage and runoff at the source of river systems and at the high altitudes which usually receive heavier rainfall. The estimated water losses are in the same range as observed for storm runoff ratios of other blanket peat covered upland catchments (Holden & Burt, 2003) and the monthly runoff versus precipitation totals shown for Moor House NNR in Evans et al. (1999). However, no data are available for an uncut catchment and therefore only the two main managements (i.e. mowing versus burning) could be compared.

Table 11 The calculated monthly percentage water loss from each sub-catchment (C = control-burnt; T = treatment-mown) at each site for each year based on monthly totals of incoming rainfall and stream flow rates. Calculations accounted for differences in sub-catchment size. Months in which management applications were carried out are highlighted in green for mowing and in brown for burning. Cells are highlighted pale blue for pre- and dark blue for post-management months where a sub-catchment showed water losses which were at least 10% greater than those from the paired sub-catchment.

Water loss (%)	Year	2012		2013		2014		2015		2016	
	Month	% loss C	% loss T	% loss C	% loss T	% loss C	% loss T	% loss C	% loss T	% loss C	% loss T
Nidderdale	1			101	103	80	64	81	64	96	72
	2			133	149	86	74	78	56	84	62
	3			103	106	61	50	81	56	83	60
	4			56	72	44	33	31	23	53	37
	5			42	47	66	45	48	37	17	13
	6			3	6	14	13	43	34	39	29
	7	119	126	18	14	11	9	16	11	34	26
	8	46	49	50	34	20	16	41	32	48	24
	9	60	60	31	23	14	9	20	16	43	19
	10	69	72	69	50	65	48	59	45	60	28
	11	82	86	71	54	73	54	80	62	93	46
	12	81	80	64	52	70	54	83	65	78	39
Mossdale	1			101	86	85	69	103	75	104	79
	2			102	91	87	71	84	64	99	78
	3			70	69	64	52	75	60	91	73
	4			54	55	48	37	53	39	64	49
	5			49	43	52	39	61	45	15	8
	6			19	14	29	20	47	35	11	7
	7	87	65	39	25	2	1	31	25	51	37
	8	47	52	69	43	39	29	62	48	74	58
	9	59	58	52	37	8	3	31	18	70	52
	10	64	67	80	60	65	48	63	49	59	40
	11	71	68	86	64	76	53	91	70	105	83
	12	85	76	80	65	79	59	100	75	84	62
Whitendale	1			150	155	60	63	84	89	82	81
	2			106	111	67	67	67	65	99	94
	3			43	43	55	55	64	64	70	66
	4			49	49	46	49	45	45	54	53
	5			46	48	41	40	54	51	31	32
	6			31	35	28	33	42	60	47	45
	7	78	86	35	35	8	11	27	26	92	112
	8	48	40	48	56	47	56	35	37	59	67
	9	68	64	51	52	34	45	32	35	48	53
	10	76	75	56	61	51	54	51	37	55	68
	11	73	72	69	86	55	55	82	87	80	91
	12	80	79	64	62	93	99	74	80	74	81

The lack of difference in water loss between the sub-catchments at Whitendale could be explained by Whitendale being the only site for which burning was rather unsuccessful in 2013 (i.e. lots of vegetation and moss layers were left intact; see Sections 3.3 and 4.2.6), which did not result in the increased area of bare/burnt ground that was observed on FI plots at Nidderdale and Mossdale (see Section 4.2.6.3). In addition both the control and mown catchments at Whitendale received some mowing before the start of the project in 2009 (see **Figure 2b**). Moreover, water table depths at Whitendale were already lowest on burnt (FI) plots before the onset of management (see **Figure 45**) and did not show any clear change over time. Importantly, the observed annual water loss reduction (as a proportion of total incoming rainfall; **Table 11**) in the streams of the mown sub-

catchments of about 20% (**Table 12**) after two management interventions (2013 & 2015) could potentially increase further, as so far only about 50% of the tall heather areas in each sub-catchment has been managed differently since 2013. Longer term monitoring over a complete management cycle would be needed to assess the full impact at the catchment-scale.

Table 12 Annual summary of the percentage of water lost in streams (relative to the total incoming rainfall) from each sub-catchment (see **Table 10** for flow volumes) and the calculated difference of water loss between mown and burnt sub-catchments (negative numbers indicate lower losses from mown compared to burnt sub-catchments) for Nidderdale (Nidd), Mossdale (Moss) and Whitendale (Whit). Calculations accounted for differences in sub-catchment size.

Nidd	%Burnt	%Mown	Mown-Burnt
2012	76	79	3
2013	62	59	-2
2014	50	39	-11
2015	55	42	-13
2016	61	38	-23
Post only	**57**	**44**	**-13**

Moss	%Burnt	%Mown	Mown-Burnt
2012	69	64	-4
2013	67	54	-13
2014	53	40	-13
2015	67	50	-17
2016	69	52	-17
Post only	**64**	**49**	**-15**

Whit	%Burnt	%Mown	Mown-Burnt
2012	71	69	-1
2013	62	66	4
2014	49	52	3
2015	55	56	2
2016	66	70	4
Post only	**58**	**61**	**3**

Whilst the observed reduction in stream water losses on mown compared to burnt sub-catchments could be linked to increased evapotranspiration from the faster regrowth of vegetation, this is unlikely to be the main reason. Firstly, vegetation cover was fairly similar between mown and burnt plots with only an initial slower heather regrowth (Section 4.2.6.3), and secondly, WTDs were higher (wetter) on mown plots (Section 4.2.7), indicating greater infiltration rates rather than greater evaporation losses. However, the implications of this are potentially wide-reaching. To illustrate their significance, the observed 20% reduction in water loss under mowing was applied across the entire Ouse catchment area (which includes both Nidderdale and Mossdale) for areas visually managed as grouse moors (using Google Earth to identify areas of rotational burning; see Appendix 4 for a more detailed methodology), assuming an average annual water loss of 60% from grouse moors under burn rotation (**Table 12**), to calculate the potential reduction in total water volume entering the Ouse. This flow volume reduction was then related to a potential decrease in river levels in York based on the available river station data on flow rates and river stage levels at Skelton (a village just upstream of York).

The river flow and stage level data from the Skelton gauge location on the River Ouse, (National River Flow Archive: https://nrfa.ceh.ac.uk/) were first used to derive a relationship between river level and peak flow rates. The best fit equation between river height and flow rate was: river height (m) = -0.000016*flow $(m^3/s)^2$ + 0.020667*flow (m^3/s); R^2 = 0.92. This relationship enabled a change in the River Ouse level to be predicted from calculated change in river flow volume due to mowing. This was calculated, assuming an approximately -20% (i.e. reduction) based on the flow reduction averages for the two Ouse catchment sites, Nidderdale and Mossdale (**Table 12**). This reduction in flow volume (i.e. water loss of 60% for burnt versus 40% for mown) was applied across all visually identified grouse moor managed areas within the Ouse catchment (18 areas of combined 389 km^2 within the total Ouse catchment of 3,397 km^2) based on rainfall amounts (adjusted for altitude). Two scenarios were considered: monthly and daily rainfall events, with daily events being the most likely to be related to large floods in York (as was the case during the severe Christmas floods in 2015). During peak rainfall events (estimated based on rainfall amounts observed during the period 2012-2016), the mowing management was estimated to cause reductions in monthly (-5.0 m^3/s) or daily (-55.6 m^3/s) flow volume at the Skelton gauge location (average peak flow volume of 380 m^3/s). These estimated reductions in flow volume, based on assumed management change from burning to mowing across the Ouse catchment, equated to possible average reductions in river levels of between 4 cm (for the monthly rainfall scenario) and 50 cm (for the daily rainfall scenario) across peak flow periods during 1982 to 2004 (**Figure 50**). Notwithstanding that flow rates - and thus river levels - depend on ground conditions, runoff and infiltration rates, which are highly variable and thus likely to differ from the 20% reduction assumed here, this modelling exercise demonstrates the potential significance for downstream flooding via effects of alternative management on catchment-scale water loss.

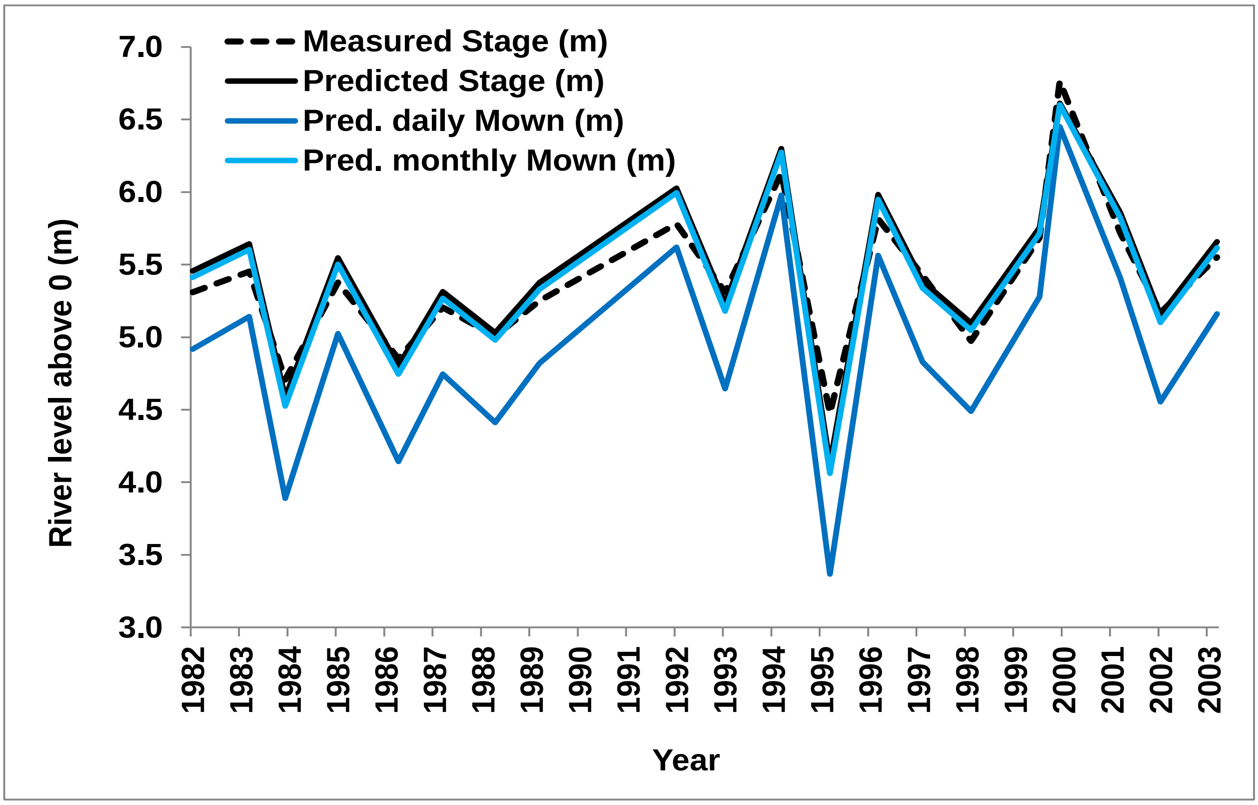


Fig. 50 Comparison of measured Ouse peak flow river stage levels (the stage is the water level above some arbitrary zero river height point) for the 20 highest events during 1982-2004 at the Skelton river gauge location, north of York (National River Flow Archive: http://nrfa.ceh.ac.uk/data/search) versus predictions (based on best fit regression of river height versus flow volumes) and for two flow reduction scenarios (considering either mean maximum monthly or daily rainfall amounts) applied across the Ouse catchment. Flow reductions were based on the averages for the Ouse catchment sites Nidderdale and Mossdale in **Table 12** (i.e. on average 20% water loss reduction from mown compared to burnt sub-catchments) and resulted in lowering of peak river levels by either 0.04 ± 0.01 m (monthly) or 0.52 ± 0.13 m (daily).

In summary, the observations of plot water table depth (Section 4.2.7), catchment stream flow (Section 4.2.8) and catchment water balance (Section 4.2.9), with the associated modelling, showed that:

- The water table depth was lower under burnt than mown plots. This is consistent with the lower surface soil moisture values in burnt than in mown plots that is reported later in the report (Section 4.5.2). These differences were greater when leaving brash, particularly at the driest site.
- The three sites differed both in the overall water table depth and in the effect of management on it, partly relating to the overall difference in rainfall.
- Stream water flows were higher in the burnt than in the mown catchments at Mossdale and Nidderdale, but not at Whitendale.
- When these water flows were combined with rainfall data to estimate the annual percentage of rainfall lost in stream water, the values were about 10% higher on burnt than mown catchments after the first management intervention at Mossdale and Nidderdale, but not at Whitendale. These differences increased to 20% higher after the second management intervention as the heather dominated area of the sub-catchment under management increased from about 25 to 50%.
- The lack of any difference between treatments at Whitendale may be due to mowing of both sub-catchments three years before the experiment started. This site also had the least successful burns.
- The 20% value of reduced stream flow was applied across all burnt areas of the Ouse catchment to model effects under flood conditions. This showed the potential for mowing to reduce a peak flood event in York by up to around 50 cm.

4.2.10 *Water quality*

Water samples were taken (see **Figure 51**) from both stream (monthly) and plot locations (3-4 times per year; see **Table A5.1** in Appendix 5) to assess changes in water quality and carbon export rates over time in relation to the possible impacts of management and climate. Flow samples were taken from the outflow of each sub-catchment V-notch weir in 250 ml bottles. Plot samples were collected overnight from within the management and slope plot water table depth locations in 50 ml syringes, which were covered to protect samples from photo-degradation. Syringes were connected to Rhizon samplers (van Walt; 0.15 µm), which were inserted vertically 10 cm into the peat, and then the plunger was pulled out to create a vacuum within the syringe and draw the water out (**Figure 51**).

Fig. 51 From left to right: Sampling soil pore water (note the covered syringe to prevent photo-degradation or freezing of water), and water samples for water quality analysis (note the wide range of colour intensity in the water samples); UV spectrophotometer; pH-meter; conductivity analyser; Inductively Coupled Plasma (ICP) for elemental analyses.

Flow and plot samples were analysed for pH and dissolved organic carbon (DOC), with samples being (re-)filtered through Rhizon samplers before DOC analysis. Stream flow sub-samples were first filtered to 0.70 µm and the residue was analysed for particulate organic carbon (POC). Moreover, flow samples from 2015 to 2016 were also assessed for the fraction of carbon which is often 'missed' due to the filter pore sizes, i.e. the fraction of DOC between Rhizon samplers (0.15 µm), the frequently used microfilters (0.45 µm) and POC (0.7 µm). The ultra-violet (UV) absorbance of the flow samples, and in 2016 of the plot samples, was measured at 254 nm, 400 nm, 465 nm and 665 nm (UV254, UV400, UV465 and UV665, respectively), which enabled calculation of the E4/E6 ratio (UV465/UV665), Hazen values (UV400 m^{-1} x 12; using the average conversion factor reported in Watts et al., 2001) and the specific ultra-violet absorbance (SUVA; UV254/DOC concentrations throughout the report are in L mg^{-1} m^{-1} unless otherwise stated). Elemental analysis was also carried out on the flow samples using an inductively coupled plasma mass spectrometer (ICP-MS) to determine the concentration of Ca, Cu, Pb, Fe, K, Mg, Na, Zn, Mn, P and Al. For more detailed methodological information on water quality see Appendix 5.

4.2.10.1 *Plot water quality*

There was no significant interaction effect on pH between the managements and time periods ($F_{3,\ 941}$ = 0.43, p = 0.73), and the time series of plot water pH (**Figure 52**) did not indicate any obvious increase in pH on burnt plots when compared to the mown or uncut plots, as was suggested by Allen (1964) based on laboratory studies. However, the EMBER project (Brown et al., 2014) also did not detect any significant pH differences between burnt and unburnt peat pore water samples, but reported lower pH in watercourses draining from burned bog catchments. Over the five year period there was a noticeable increase in pH values (about 1 unit) across all sites, which was most pronounced at Nidderdale (**Figure 52**). Overall, all three sites showed very similar average pH values (3.5 pre- and 4.1 post-management) which are around the anticipated threshold of pH >3.8 for a recovering blanket bog (Anderson et al., 1997). Overall, the linear mixed-effects model revealed that the month of sampling, mean soil temperature (coefficient: Tsoil = 0.25; $Tsoil^2$ = -0.01) and total rainfall (coefficient: 0.003) in the four weeks prior to sampling significantly affected the pH (p< 0.001 for all). In particular, June 2014 samples showed substantially lower pH values at all sites; this coincided with a warm period combined with very low monthly rainfall (**Figure 10**) and low WTDs (**Figure 45**).

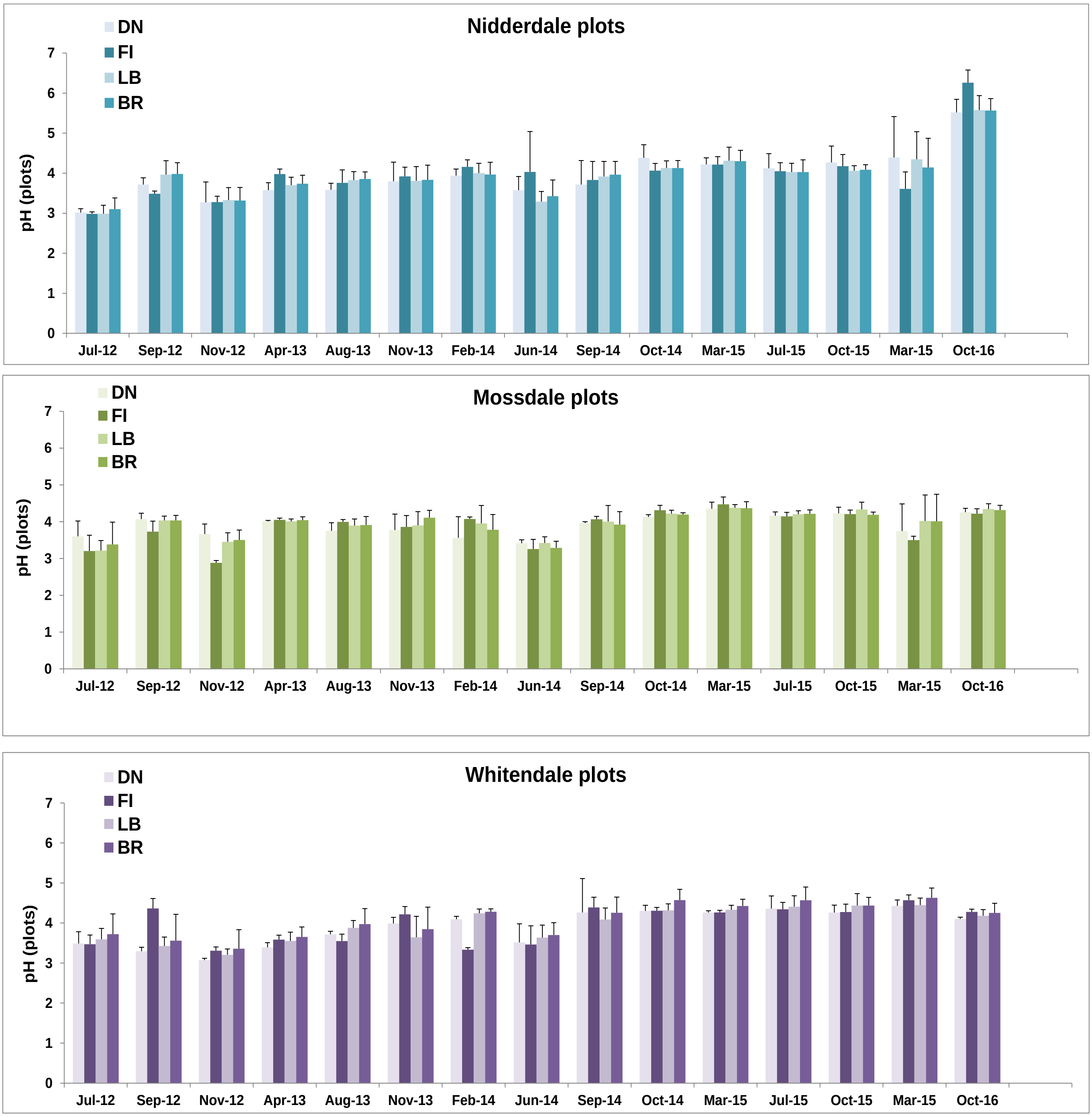


Fig. 52 Mean (+ standard deviation) pH values for peat pore water samples (taken with a Rhizon sampler) at the three sites (**top**: Nidderdale; **middle**: Mossdale; **bottom**: Whitendale) throughout the pre- (2012) and post- (2013-2016) management periods for the major plot managements (uncut (DN), n = 4; burnt (FI), n = 4; combined *Sphagnum* treatments for mown with left brash (LB), n= 8, and with brash removed (BR), n = 8).

Pore water pH in the slope plots (**Figure 53**), which had varying vegetation but were mostly covered by dense heather or sedge, showed a similar pattern to the plot water pH values, with an overall pH increase over five years by about half to one pH unit. This is similar to short-term (i.e. a peak rather than an overall trend) changes reported by Evans et al. (2017a) for streams in a UK upland peat area.

Fig. 53 Mean (+ standard deviation) pH values for peat pore water samples (taken with a Rhizon sampler) at the three sites (**top**: Nidderdale; **middle**: Mossdale; **bottom**: Whitendale) throughout the pre- (2012) and post- (2013-2016) management periods for the slope locations (n = 18), which were not managed as part of the experiment and had a mixture of tall vegetation and previously burnt areas within both, the burnt (Control) and the mown (Treatment) catchments.

The linear mixed-effects model indicated no significant differences in pH between slope samples from burnt and mown sub-catchments in the pre- or post-management periods ($F_{1,\ 1473} = 0.08$, $p = 0.77$). As with the plot pH, the month of sampling, mean soil temperature (coefficient: Tsoil = 0.18; $Tsoil^2 = -0.01$) and total rainfall (coefficient: 0.008) in the four weeks prior to sampling had a highly significant ($p< 0.002$ for all) effect on pH but there were no significant differences between sites ($F_{2,\ 6} = 2.15$, $p = 0.20$). A linear regression analysis revealed that the overall pH increase over time was significant ($p< 0.001$) for both, plots and slopes, although the R^2 value was only 0.3. Unfortunately, there are not many data available for comparable small blanket bog catchment streams. However, Evans et al. (2005; cf. Fig. 3 & 5) and Evans et al. (2017a; cf. Fig. 2) report similar changes of up to 0.5 pH units.

The linear mixed-effects models for plot pore water DOC concentrations (**Figure 54**) revealed overall site differences ($F_{2,\ 9}$ = 20.92, $p < 0.001$), with Mossdale having lower concentrations than the other two sites (53.3 mg L^{-1} compared to 79.0 and 70.3 mg L^{-1} for Nidderdale and Whitendale, respectively). Although there were no significant differences between periods overall ($p > 0.3$), there was an interaction between management and site ($F_{6,\ 955}$ = 7.46, $p < 0.001$) which was largely attributable to the DOC concentrations at Nidderdale being higher on LB plots (79 mg L^{-1}) than on BR (58 mg L^{-1}) and FI (47 mg L^{-1}) plots ($p < 0.001$ for both) during the post-management period, but this difference in DOC was similar in the pre-management period (100, 72 and 76 mg L^{-1}, respectively). Higher soil temperatures, averaged over the four weeks prior to sampling plot soil water, significantly increased DOC concentrations (coefficient: 0.07; F_1, 537 = 6.2, $p < 0.05$) whereas the percentage of heather negatively influenced DOC concentrations (coefficient: -0.002; $F_{1,\ 852}$ =7.50, p = 0.006). The month of measurement also significantly affected DOC concentrations ($F_{8,\ 761}$ = 23.1, $p < 0.001$).

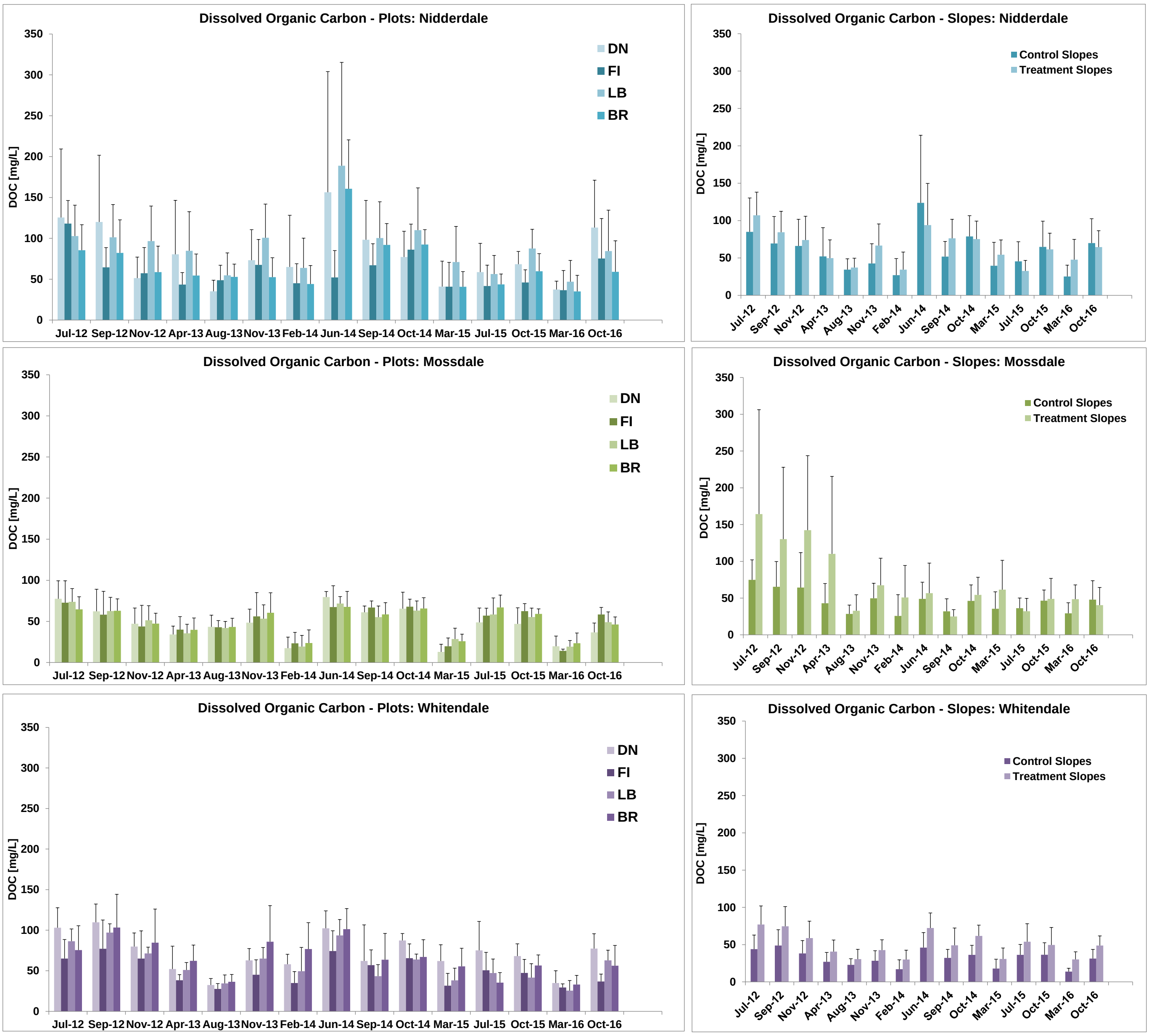


Fig. 54 Dissolved organic carbon (DOC) concentrations (mean + standard deviation) in pore water samples (taken with a Rhizon sampler) for plot (**left**) and slope (**right**) locations from the three sites (**top**: Nidderdale; **middle**: Mossdale; **bottom**: Whitendale) throughout the pre- (2012) and post- (2013-2016) management periods. Plot averages are shown for the major managements (uncut (DN), n = 4; burnt (FI), n = 4; combined *Sphagnum* treatments for mown with left brash (LB), n= 8, and with brash removed (BR), n = 8). The slope locations (n = 18) were not managed as part of the experiment and had a mixture of tall vegetation and previously burnt areas.

For the slope pore water samples, there was a significant ($F_{2, 13}$ = 7.92, p = 0.006) site effect, with decreasing mean DOC concentrations from Mossdale (76.7 mg L^{-1}) to Nidderdale (69.4 mg L^{-1}) to Whitendale (47.0 mg L^{-1}), and a weak ($F_{1, 1441}$ = 13.64, p = 0.03) but no meaningful catchment ($F_{1, 1443}$ = 77.11, p < 0.001) and interaction effect between periods and sub-catchments (i.e. as DOC slope concentrations were higher in mown than burnt sub-catchments both pre- and post-management; see **Figure 54** right hand panels). However, the mean soil temperature (and its squared term) in the four weeks prior to sampling (coefficients: 0.26 and -0.01, respectively) were highly significant (p< 0.001), and the heather cover (positive coefficient: 0.002) was weakly significant (p< 0.05) in explaining DOC differences. Across both plots and slopes, highest slope DOC concentrations during the post-period were measured during the exceptionally dry and warm June 2014 sampling, which is likely related to the observed decrease in pH (see **Figures 52** and **53**), although the mechanisms can be complex (Clark et al., 2012), reflecting chemical and concentration changes after rewetting. The lack of any significant effect of management after intervention is consistent with the findings of Worrall et al. (2013), who also observed a lack in significant differences between DOC concentrations on mown, uncut or burnt plots.

The ranges of peat pore water DOC concentrations (**Figure 54**) during the five year period were well within the reported range for plots in UK upland blanket bogs ranging from 5 to 120 mg L^{-1} (Armstrong et al., 2012) and showed a clear pattern of overall declining DOC concentrations at both plots (78 to 58 mg L^{-1}) and slopes (82 to 46 mg L^{-1}) during the post- compared to the pre-management period; although slope measurements are not directly related to management it is indirectly, as the slope areas are hydrologically connected to a differently managed catchment. The only major environmental changes recorded during the monitoring period were increasing soil temperatures (by about 1^{o}C; see **Table 1**) and increasing pH (by about 1 unit; see **Figures 52** and **53**). This negative pH link to DOC is interesting, as previous work (e.g. Dawson et al., 2009) has shown the opposite (i.e. positive) correlation of pH to DOC concentration.

There were significant differences in the water quality parameters between plot pore water samples collected in March and October 2016, but there were no management differences in any of the investigated parameters (p > 0.05; **Figure 55**). The E4/E6 ratio was significantly lower at Mossdale in March than at Nidderdale (p = 0.03) and in October than at Whitendale (p = 0.02), UV254 was significantly lower only in October at Mossdale than at Nidderdale (p< 0.01) and Whitendale (p< 0.05), Hazen (or UV400) was significantly lower in both months at Mossdale than at Nidderdale or Whitendale (p< 0.05 for both), and SUVA was lower at Nidderdale in March than at Mossdale (p = 0.004) and in October than at Whitendale (p = 0.04). Therefore, the wettest site Mossdale showed overall lowest E4/E6 (2.77), UV254 (209 Au m^{-1}) and Hazen (437) plot pore water values, whilst the driest site Nidderdale showed lowest SUVA (5.28 L mg^{-1} m^{-1}).

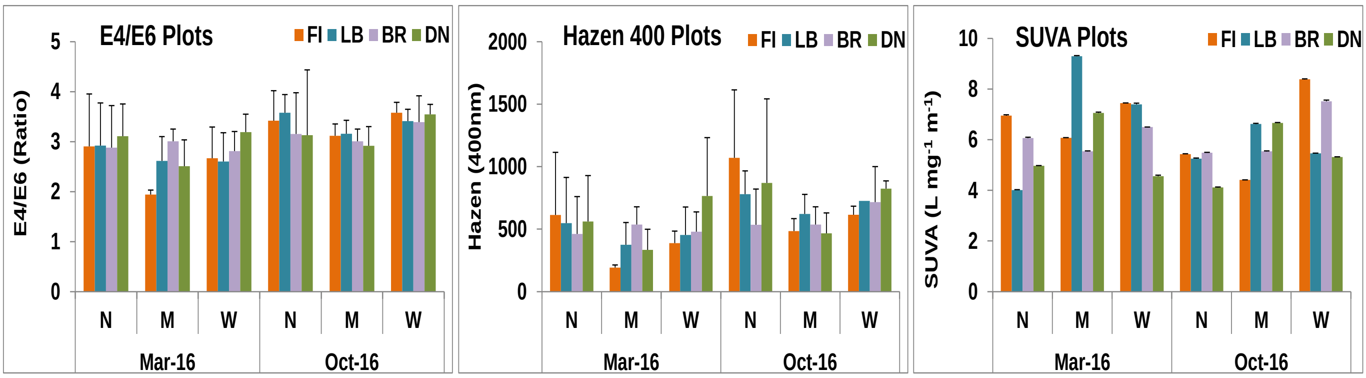


Fig. 55 Water quality means (+ standard deviation) for plot locations measured as E4/E6 ratios (**left**), Hazen (**middle**) and SUVA (**right**) for peat pore water samples (taken with a Rhizon sampler) across the three sites (N = Nidderdale, M = Mossdale, W = Whitendale) during 2016 (post-management) for the major plot managements (Burnt (FI), n = 4; combined *Sphagnum* treatments for mown with left brash (LB), n= 8, and with brash removed (BR), n = 8; uncut (DN), n = 4).

The water quality parameters (**Figure 56**) for samples from the slope locations showed very similar ranges to those observed at the plot locations. There were strong site effects on the E4/E6 ratio (overall mean = 3.05) and Hazen (overall mean = 557.9) ($p < 0.001$ for both) and a weak site effect ($p < 0.03$) on SUVA (overall mean = 6.08 L mg^{-1} m^{-1}); Whitendale had lowest E4/E6 ratios (2.62) and Hazen values (375), and SUVA was higher at Mossdale than at Whitendale. Therefore, site differences for slopes did not reflect those obtained from plots. There were also significant catchment differences ($p < 0.05$) for all parameters in both months except for SUVA in March, with burnt catchments showing lower values than mown catchments for E4/E6 (2.87 versus 3.26), Hazen (483 vs 639) and UV254 (215 vs 286). However, although no pre-management data were available, these differences were unlikely to be related to previous management given the lack of any coherent management for slope locations.

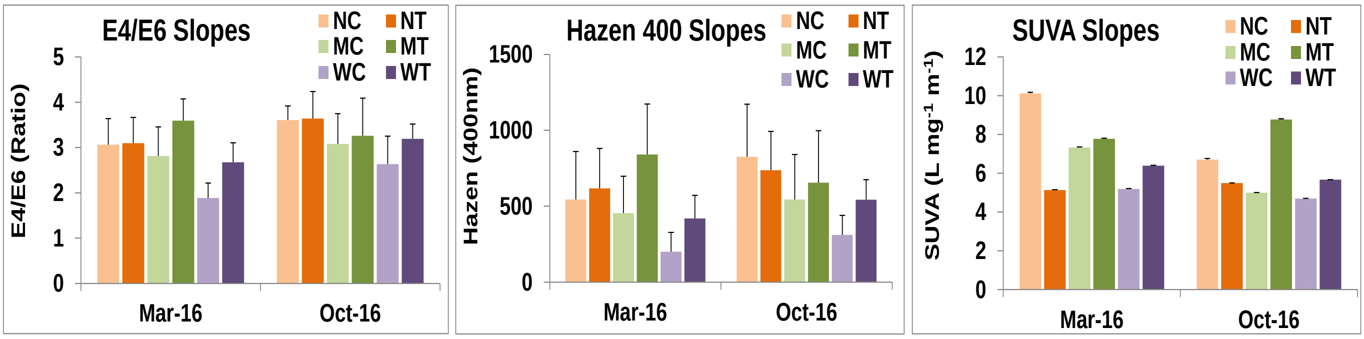


Fig. 56 Water quality means (+ standard deviation) for slope locations measured as E4/E6 ratios (**left**), Hazen (**middle**) and SUVA (**right**) for peat pore water samples (taken with a Rhizon sampler) across the three sites (N = Nidderdale, M = Mossdale, W = Whitendale) during 2016 (post-management). The slope locations (n = 18) were not managed as part of the experiment and were a mix of heather regrowth stages on previously burnt areas within the managed sub-catchments (C = burning, T = mowing).

The plot and slope water quality analyses indicated clear site differences, highlighting the need to consider individual sites in the context of generic differences in climate and vegetation. However, site differences were not the same between plots and slopes. Although short- and long-term, and site-specific, differences in water quality parameters have been reported for DOC, Hazen, E4/E6 and SUVA (e.g. Dielman et al., 2016; Peacock et al., 2014; Wang et al., 2013; Watts et al., 2001), the underpinning processes are still largely unknown. Some of the observed variability is most likely explained by interactions between temperature and water table changes as observed by Dielman et al. (2016) for DOC and SUVA. Notably, the Peacock et al. (2014; cf. Fig. 7) study observed similar ranges of SUVA (i.e. low median but high upper quartile and maximum range) to our study (i.e. **Figures 60** & **61**).

Moreover, data on vegetation and location specific impacts on DOC and water quality are still sparse. Ritson et al. (2014) indicate higher leached DOC concentrations under heather compared to *Sphagnum*-dominated areas. Armstrong et al. (2012) showed that DOC was highest in ditches amongst heather- as opposed to sedge-dominated vegetation, yet the median values were nearly identical for pore water and there was only one sampling event per week over three weeks in the summer of one year. For example, the plot samples in Armstrong et al. (2012) were not significantly different between *Calluna* and sedge. Only the drain (ditch) samples showed a difference, but the vegetation versus erosion impact for those was not discussed or measured. The present study (when using sub-catchment averages) also showed no significant difference for DOC but indicated a possible opposite effect, with a (so far non-significant) negative relationship of DOC concentrations from the plots (but not slopes) with heather and a positive relationship with sedge (**Figure 57a**). These data were collected at a similarly low sampling frequency to those in Armstrong et al. (2012) but spanned a wider period during 2016. Interestingly, they do agree with the negative correlation which was observed between all 15 DOC plot-sampling times and heather cover across the entire 5-year monitoring period (see above). Moreover, as an average per sub-catchment, SUVA seemed to increase with increasing sedge (which was nearly exclusively *Eriophorum* spp.) (**Figure 57a**). However, linear regression for the average plot-level management showed a significantly positive SUVA correlation with *Sphagnum* cover ($p < 0.01$) and a negative one with heather cover ($p < 0.05$) and on

individual slope locations a significant (p< 0.001) positive correlation with *Sphagnum* and sedge cover (**Figure 57b**), but correlation coefficients were again very low.

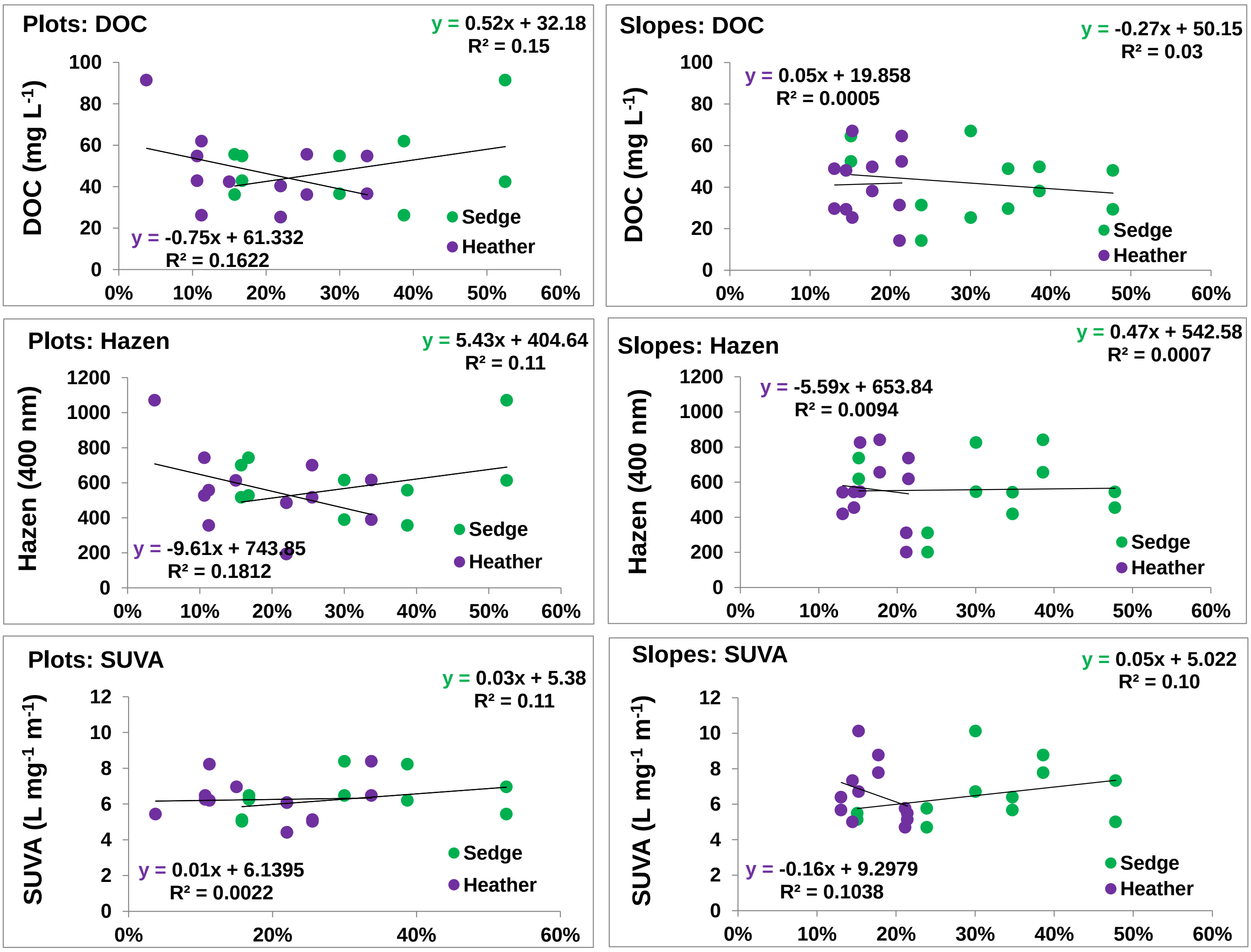


Fig. 57a Water quality averages (per sub-catchment) in 2016 (post-management) for plots (**left**) and slope locations (**right**) at the three sites measured as DOC (**top**), Hazen, (**middle**) and SUVA (**bottom**) versus the average cover (%) of either heather (*Calluna vulgaris*; purple) or sedge (*Eriophorum* spp.; green) at the locations. Note the slope locations were not managed as part of the experiment and were a mix of heather regrowth stages on previously burnt areas within the managed sub-catchments. Linear regression equations (non-significant) are provided for each graph (same colour code as for vegetation).

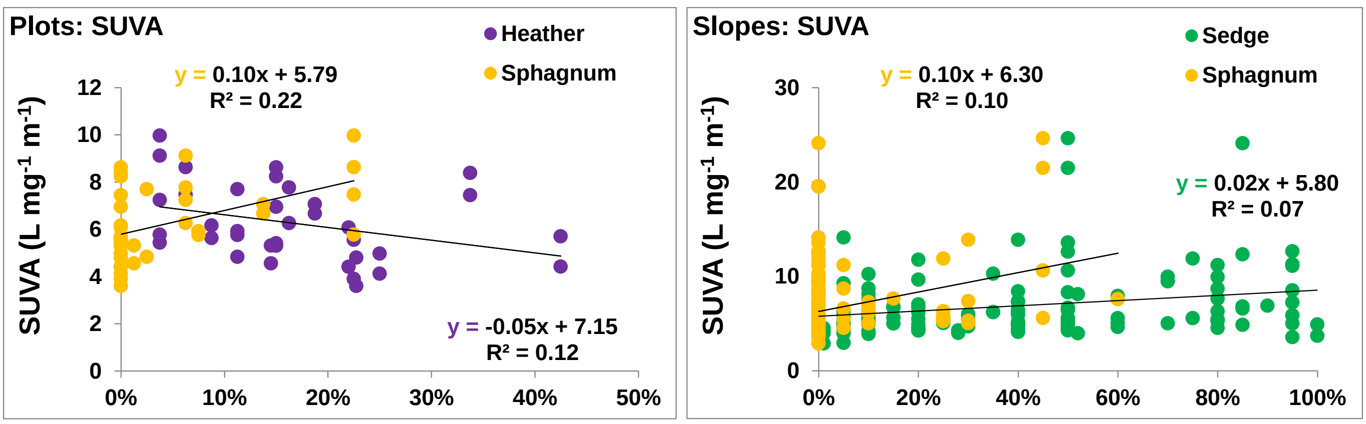


Fig. 57b Water sample SUVA values in 2016 (post-management) as the mean per management for plots (**left**) and the individual slope locations (**right**) at the three sites against heather (purple), *Sphagnum* (orange) or sedge (green) species cover at the sample locations. Note different axes scales, and slope locations were not experimentally managed and were a mix of heather re-growth stages on previously burnt areas within the managed sub-catchments. Linear regression equations provided for each graph (same colour code as for vegetation) are significant (p< 0.001).

Fenner et al. (2004) also showed DOC impacts by *Sphagnum* and Vestgarden et al. (2010) showed higher DOC for *Sphagnum* in 10 cm peat depth (and discussed a potential concentration effect). Clearly more data on vegetation impacts on water quality are needed, ideally under controlled mesocosm conditions (as done by Dielman et al., 2016), to reveal the underpinning processes in relation to possible management induced changes via shifts in vegetation composition.

In addition to root aeration impacts (i.e. O_2 transporting aerenchyma in sedges) on decomposition affecting sulphate (SO_4) production (Lamers et al., 2013), and thus potentially also DOC, predicted changes in peatland acidity, specifically in relation to recovery from acidification (Dawson et al., 2009; Monteith et al., 2014) and low peat moisture or drought conditions (Clark et al., 2006), are also important when considering water quality and DOC. Interestingly, the observed pH change of about one unit over only five years could reflect a noisy but upward long-term trend in stream pH as was predicted by Evans et al. (2005) until 2020 in response to recovery from acid deposition. Importantly, any changes should be observed first in the peat pore water itself, which feeds into the streams, as the signal is diluted across vegetation and landscape (due to variations in topography, wetness and soil types). Contrary to findings by Worrall et al. (2013), this study did not observe a reduction in E4/E6 ratios on burnt plots compared to mown or uncut plots (**Figure 55**), although Worrall et al. observed this reduction only on fresh burns and 5-year old burns but not on 1-year old burns. However, they did acknowledge that without available pre-management data any differences could have been unrelated to management.

4.2.10.2 *Stream water quality*

Monthly stream samples (for dates see **Table A5.2** in Appendix 5) allowed comparison of sites and their paired sub-catchments (burnt and mown) in water chemistry, as well as changes over time. Stream pH increased significantly (in a mixed effect model) compared to the pre-management change year ($p < 0.01$) at all sites in the last two post-management years (**Figure 58**). A similar increase by 0.5 units was also observed at the plot and slope locations (**Figure 52** and **53**, respectively). Nidderdale had significantly lower pH values ($F_{2,\ 108} = 13.22$, $p < 0.001$) than the other two sites. Although there was a significant interaction between site and management ($F_{2,\ 156} = 11.14$, $p < 0.001$), there was not between period and management ($F_{1,\ 156} = 0.07$, $p = 0.79$) indicating that there was no management effect once pre-period differences were taken into account across the three sites, especially at Whitendale. Whilst the EMBER study (Brown et al., 2013) reported similar pH values for similar sized sub-catchments (about 1 km^2), they also reported lower pH values in burnt compared to unburnt catchment rivers. Crucially, the EMBER study was based on a space for time substitution approach (i.e. without including a pre-management period), assuming that observed differences across differently managed sites reflect only management impacts (e.g. ignoring differences in elevation, rainfall and temperature). Differences in flow water pH across the time series in this study highlight the need to consider a pre-management period to capture differences unrelated to management as part of a BACI experimental design (see Schwarz, 2015), as otherwise differences occurring by chance could be interpreted as management impacts.

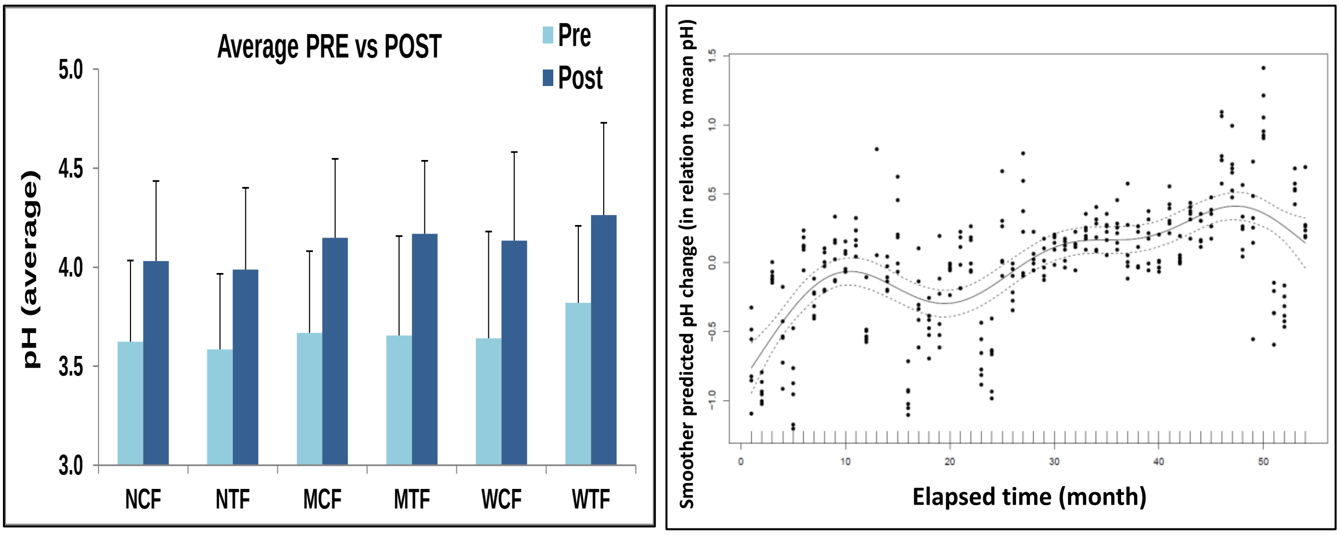


Fig. 58 Average (with standard deviation) pH values for the pre- (2012-2013) and post- (2013-2016) management periods at the flow weirs at Nidderdale (NF), Mossdale (MF) and Whitendale (WF) for the burnt (C) and mown (T) sub-catchments (**left**). A generalised additive model using all combined pH data together with mean air temperature and total rainfall amounts four weeks before pH measurements (**right**) provided a best estimate smoothing line (± 95% confidence intervals) for the long term pH (Y-axis shown as pH unit change in relation to mean pH) trend (elapsed time in months).

The stream flow conductivity (**Figure 59**, below) showed similar values to those reported in the literature for the main upland river Nidd catchment as reported by Palmer et al. (2016) with a median of 46 $\mu S\ cm^{-1}$ (range of 22 – 379 $\mu S\ cm^{-1}$). Whitendale had a mean conductivity of 54 $\mu S\ cm^{-1}$ overall which was significantly higher ($F_{2,\ 103} = 8.08$, $p < 0.001$) than that at the other two sites (Nidd: 52 $\mu S\ cm^{-1}$; Moss: 49 $\mu S\ cm^{-1}$). As for pH, although there was a significant interaction between site and management ($F_{2,\ 154} = 27.15$, $p < 0.001$), there was not between period and management ($F_{1,\ 157} = 0.0003$, $p = 0.99$) and as for pH differences were reflected pre-management and related to site rather than to management. Overall there was an indication of a seasonal pattern, with peaks tending to occur in late winter/early spring, with the maximum values observed in 2014 and 2015. However, low flow volumes during these generally dry periods (see **Table 10**) most likely explain these maxima rather than overall climatic conditions, as neither mean temperature nor total rainfall during the four weeks prior to sampling were retained in a linear mixed-effects model.

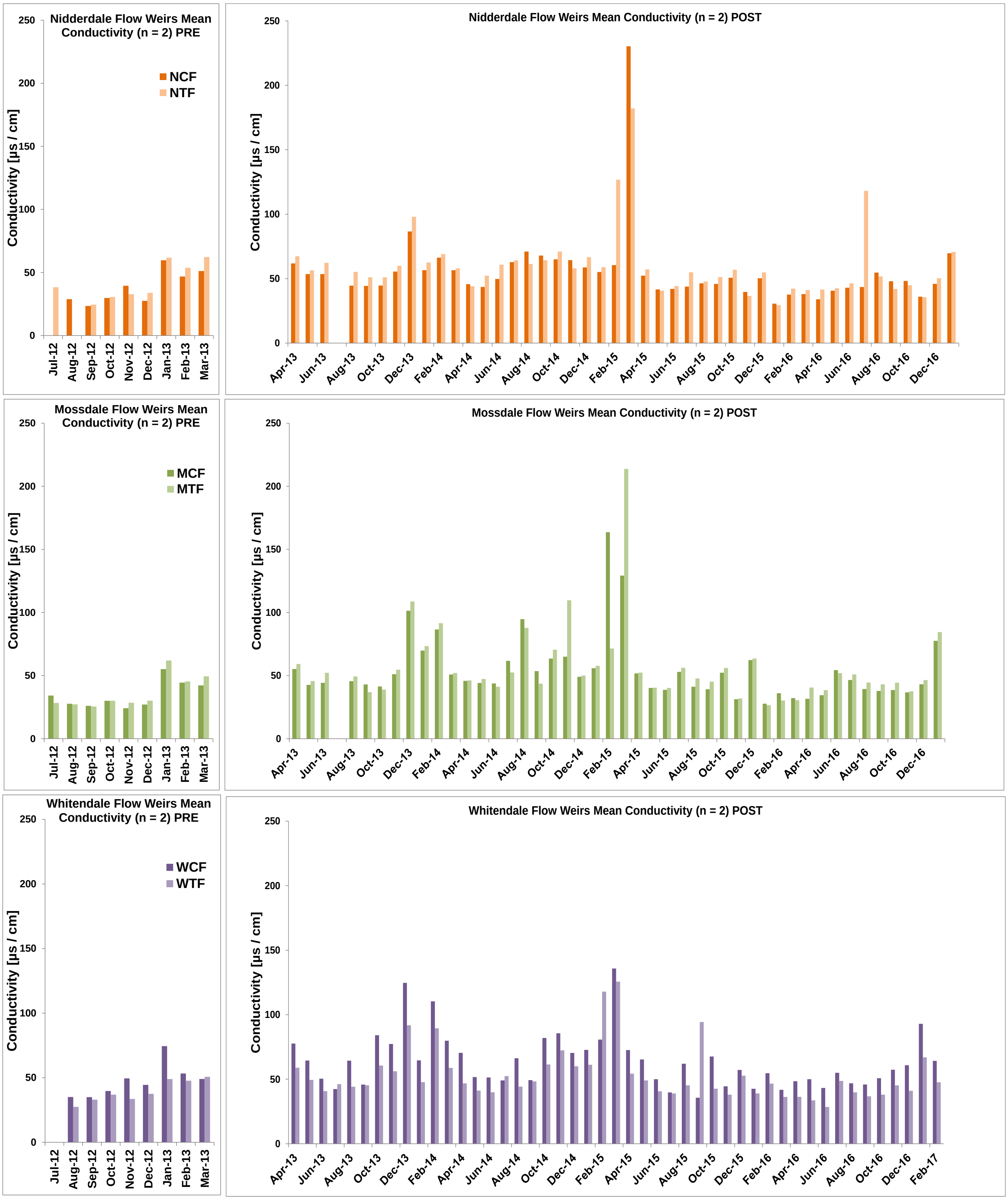


Fig. 59 Conductivity in streams for the pre- (**left: 2012-2013**) and post- (**right: 2013-2016**) management period at Nidderdale (N) (**top**), Mossdale (M) (**middle**) and Whitendale (W) (**bottom**) for the control (C, burnt) and treatment (T, mown) sub-catchment flow (F) samples.

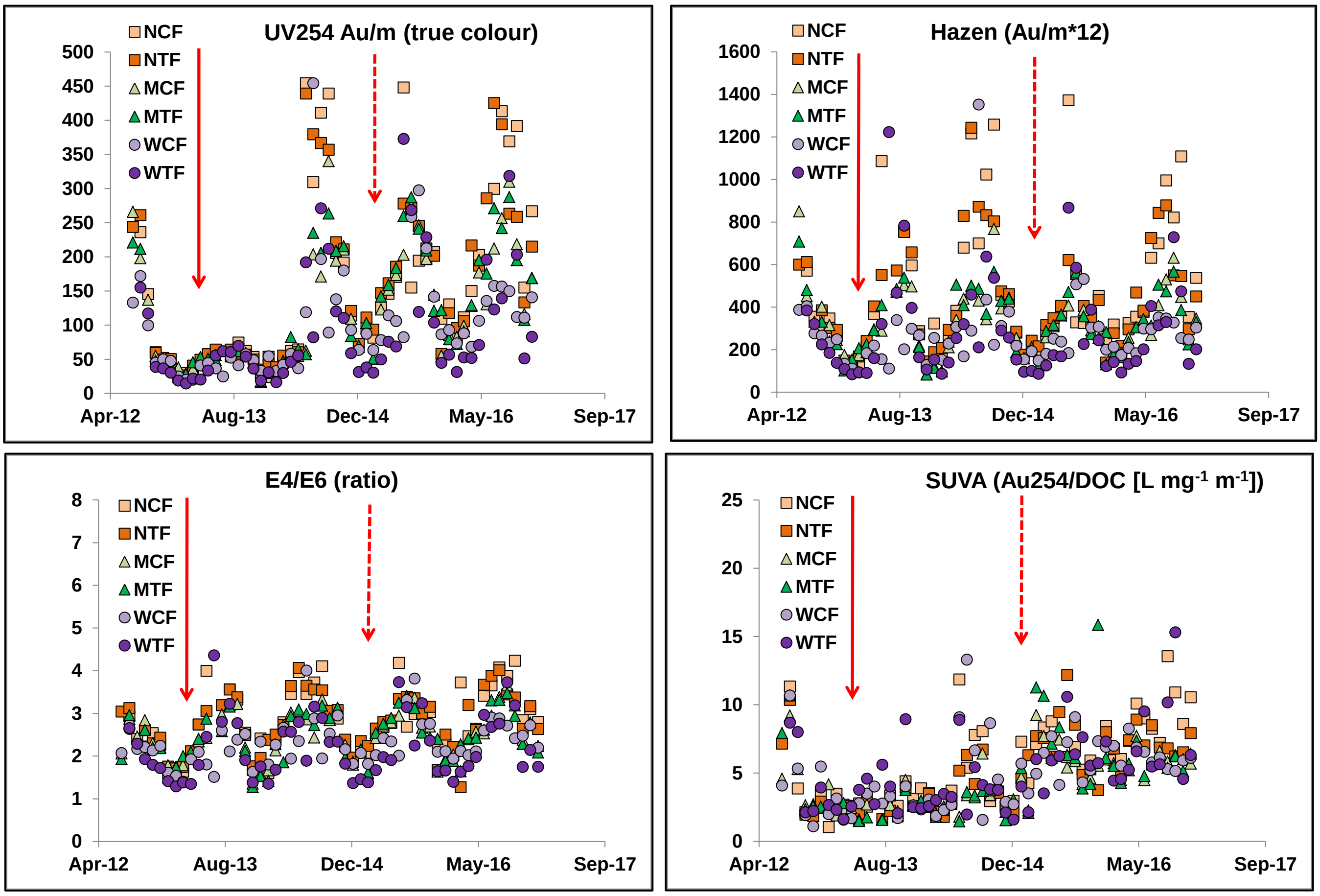


Fig. 60 Water quality measures from flow weir water samples between July 2012 and December 2016. Absorbance (Au/m) at 254 nm (**top left**), Hazen (**top right**), the absorbance 465nm/665nm (E4/E6) ratio (**bottom left**) and SUVA (L mg^{-1} m^{-1}) values (**bottom right**) are shown for each of the three sites (N = Nidderdale, M = Mossdale, W = Whitendale) for the stream flow in burnt control (CF) and mown treatment (TF) sub-catchments. The solid arrows indicate the onset of different management in March 2013; the dashed arrows indicate the 2nd management in March 2015. Note that the Hazen conversion of absorbance at 400 nm used the average factor (i.e. UV400*12) reported for Yorkshire catchments in Watts et al. (2001). Zero values due to missing data (snow or no flow) and one very high value, NCF (July-15) = 37.05, was removed.

There was clear seasonal (summer peaks) and inter-annual variation for the water quality UV spectra parameter measurements in the monthly flow samples (**Figure 60** above); UV254 and SUVA were substantially lower in 2013 compared to all other years, whereas Hazen values and E4/E6 ratios displayed a consistent annual pattern throughout the entire monitoring period. On average, UV254 values, Hazen values and E4/E6 ratios decreased significantly ($p < 0.001$) from Nidderdale (158±120; 454±280; 2.78±0.69, respectively) to Mossdale (122±76; 325±152; 2.47±0.56, respectively) to Whitendale (95±79; 276±201; 2.25±0.62, respectively). UV parameters were similar to those reported in the literature for pore and ditch water (Peacock et al., 2014), but slightly higher than data from main streams (Peacock et al., 2015) or reservoirs (Watts et al., 2001), most likely indicating a river dilution effect. Mean SUVA values (**Figure 60**) were statistically the same across the three sites (decreasing from Nidderdale: 5.8±4.3 L mg^{-1} m^{-1} to Whitendale: 5.0±2.7 L mg^{-1} m^{-1} to Mossdale: 4.6±2.5 L mg^{-1} m^{-1}). These bog stream average values were slightly higher as the 4.6 L mg^{-1} reported in the literature for the main river Nidd (Palmer et al., 2016), although this is largely due to higher values in the warmer years of 2015 and 2016 (see **Table 1** for climate data). Higher SUVA values indicate that a larger percentage of the DOC is aromatic in nature (Weishaar et al., 2003), meaning more humic and fulvic acids are present (Edzwald, 1993). Generally this is attributed to phenols linked to *Sphagnum* (e.g. Min et al., 2015), which have been shown to increase recalcitrance to decomposition (Marschner & Kalbitz, 2003). However, this SUVA increase could also relate to increased aromatics due to charcoal from burning (which was evident across all three sites for an extended period of time, see Section 4.3.3), with lowest values at the least modified site (Mossdale), which should be investigated further.

There were climatic impacts on the water parameters measured in the monthly stream flow samples: the month of measurement had a significant impact on UV254 ($F_{11, 36}$ = 11.0, p< 0.001), whilst soil temperature had a positive effect on the E4/E6 ratio and Hazen values (coefficient: 0.1 for both; $F_{1, 50}$ = 300.6, p< 0.001 and $F_{1, 52}$ = 294.7, p< 0.001, respectively), and total rainfall in the four weeks prior to sampling had a negative impact (coefficient: -0.001 for both; $F_{1, 57}$ = 24.4, p< 0.001 and $F_{1, 60}$ = 19.5, p< 0.001, respectively) .

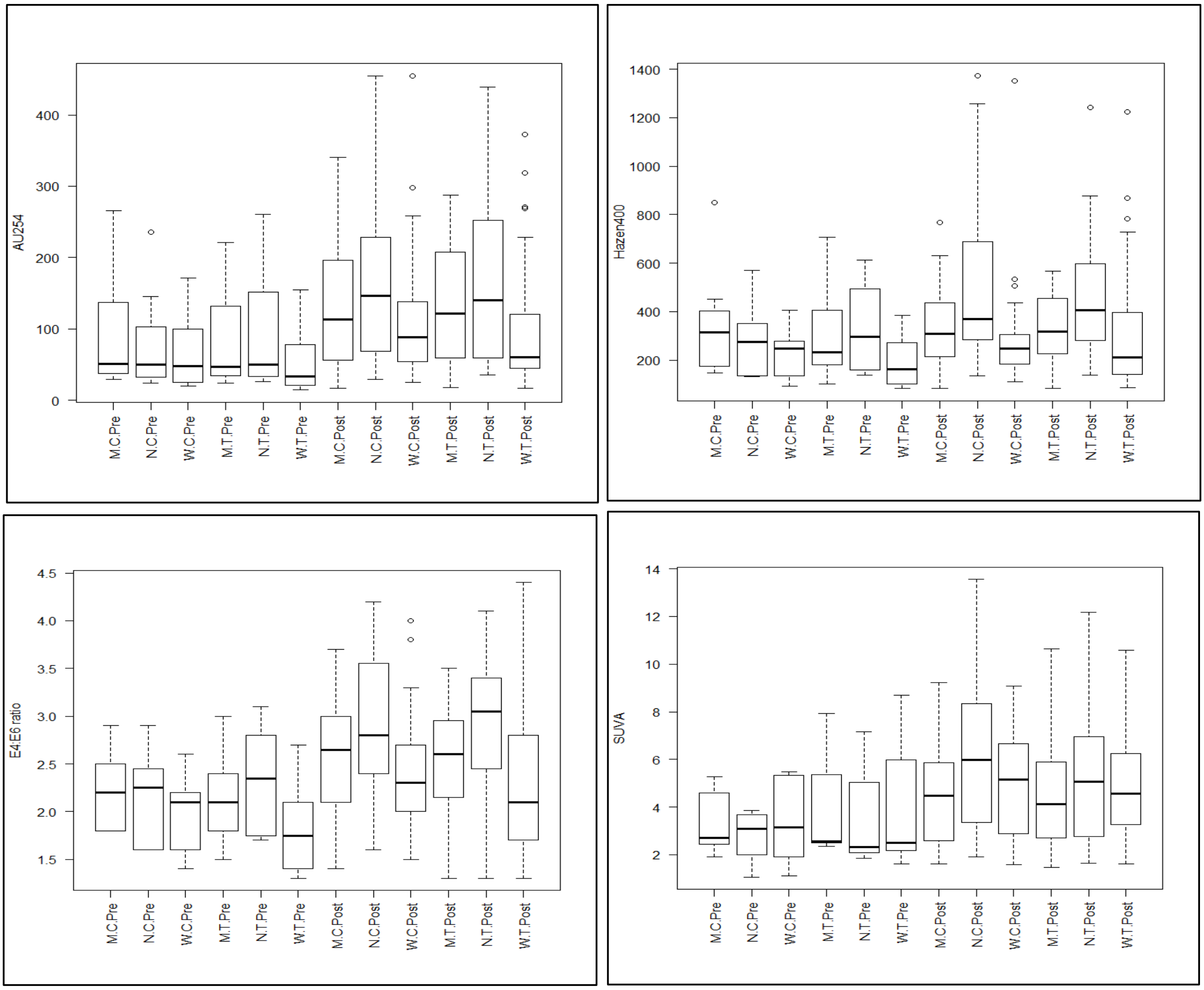


Fig. 61 Water quality measures for monthly flow weir samples (**clockwise from top left**: UV254 (Au/m), Hazen (Au/m*12), SUVA (L mg^{-1} m^{-1}) and E4/E6 ratio) from the burnt control (C) and mown treatment (T) sub-catchments at each site (N = Nidderdale, M = Mossdale, W = Whitendale) during the pre- (2012) and post-management (2013-2016) periods. The box midline indicates median, the box edges indicate the interquartile range and the points indicate data greater or less than 1.5 times the interquartile range.

There was no significant interaction between management and time period for any parameter (see **Figure 61** above), although the E4/E6 ratio was significantly higher overall in post- than pre-period ($F_{1, 22}$ = 5.37, p = 0.03) and, across both periods, UV254 was significantly greater in the burnt than mown sub-catchment at Whitendale ($F_{2, 154}$ = 5.55, p< 0.01) and SUVA was significantly greater in the burnt than mown sub-catchment at Nidderdale ($F_{2, 151}$ = 3.03, p< 0.05). However, any difference in stream water chemistry related to the management interventions would be diluted by the overall catchment signal containing previously burnt areas in both the mown and burnt sub-catchments (i.e. from areas burnt before the onset of the project). Moreover, the possible delay in filtration rates from plots through the peat into the stream would likely be more than five years (pers. comm. Andrew Walker, Yorkshire Water).

As previously observed for Hazen values during dry spells (Watts et al., 2001), the dry 2013 period in this study resulted in lower values for most parameters (**Figure 60**); in particular, UV254 was surprisingly low throughout 2013 (**Figure 62**). Interestingly, Yorkshire Water measurements for reservoir water quality in the Nidderdale area (pers. comm. Jenny Banks; Yorkshire Water) also showed a considerable drop in UV254 values in 2013, which our analysis suggests is climate related. Whereas 2013 was the brightest and driest year overall following a very wet autumn/winter (**Table 1**) with highest summer temperatures (**Figure 10**) and generally low water tables (**Figure 44**) and low flow rates (**Table 10**), 2015 and 2016 showed lower summer peak temperatures yet high mean soil temperatures overall together with high light (PAR) levels (**Table 1**; **Figure 10**) and slightly higher water tables (**Figure 44**). Interestingly, these differences in climatic conditions related to observed differences (in this study) in carbon cycling with higher methane emissions (**Table 16** see Section 4.2.15) in the last two years, particularly in 2016, which also showed lowest carbon use efficiencies (**Table 15** see Section 4.2.14), as well as the lowest UV254 and SUVA values in 2013 but highest SUVA values in 2015 and 2016 (**Figure 60**). Therefore, a combination of both changes in carbon input and decomposition processes unrelated to recent management, but related to climate and vegetation (as seen in **Figure 57**) seems a likely cause for the increased SUVA concentrations in 2015 and 2016 and warrants further investigation in field and laboratory studies. Notably, this increase in SUVA coincided with an observed sharp increase in methane emissions in 2015 and 2016 (see Section 4.2.15), suggesting a direct (most likely microbial) C cycle link.

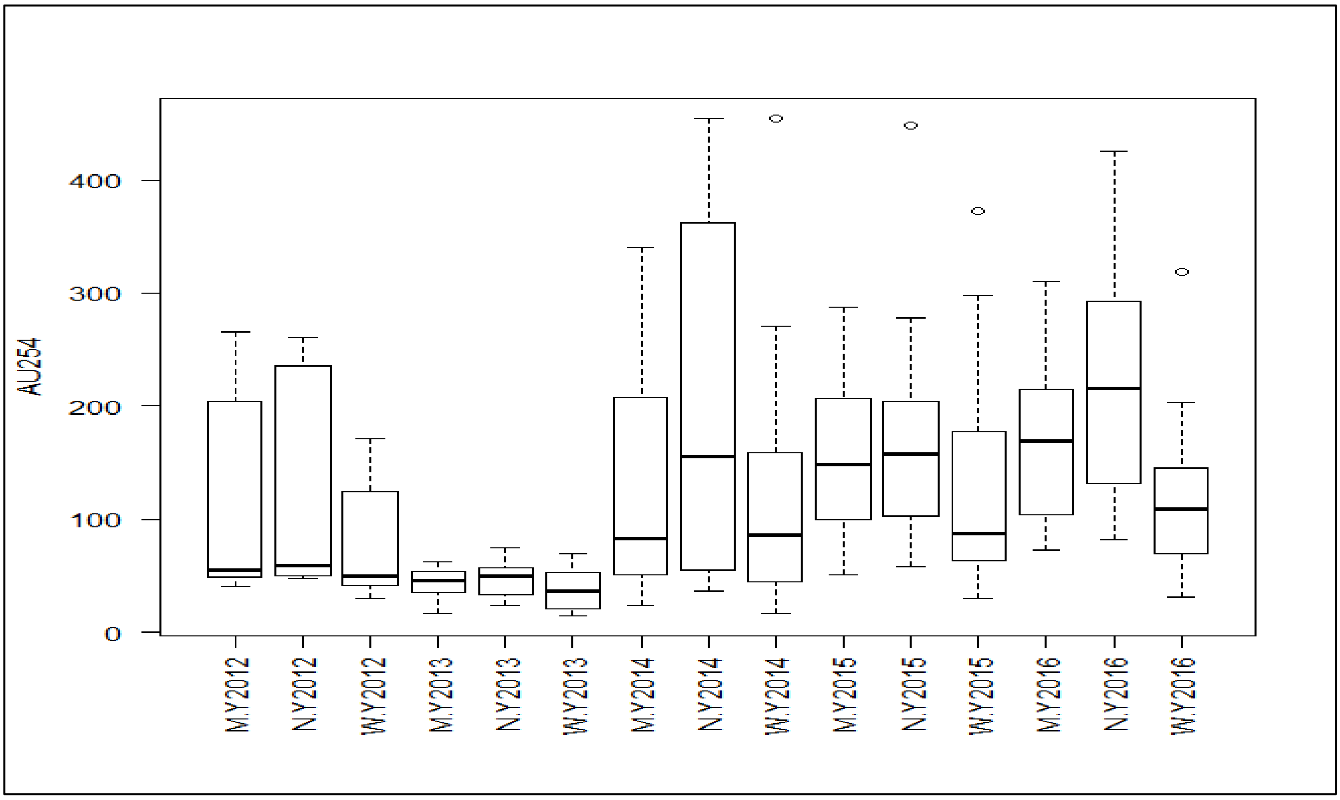


Fig. 62 Annual UV254 nm (in Au/m) absorption ranges for monthly flow weir samples for Nidderdale (N), Mossdale (M) and Whitendale (W) (with the control and treatment sub-catchments combined) over the experimental period. The onset of different management was from April 2013. The box midline indicates the median, the box edges indicate the interquartile range and the points indicate data greater or less than 1.5 times the interquartile range.

Flow samples were also analysed for elemental composition using an Inductively Coupled Plasma (ICP) Mass Spectrometer (**Figure 63**). Overall, the mean concentrations (all in ppm) across all sites and sub-catchments were: Zn (0.05); Na (3.09); K (0.57); Mg (0.50); Fe (1.67); Cu (0.01); Pb (0.01); Ca (0.91); Al (0.23); Mn (0.08) and P (0.03), all similar to ranges reported for bog catchment streams by Verry (1975). Month of measurement significantly affected Mg and K (both $p < 0.05$), Ca ($p < 0.01$), and Fe and Al (both $p < 0.001$). The concentration of K was lower in the combined post-management compared to the pre-management period ($p < 0.01$), but Zn, Fe (both $p < 0.01$), Mg and Na (both $p < 0.05$) were higher post-management (**Figure 63**).

Fig. 63 Elemental concentrations in the monthly stream flow samples for the three sites: Nidderdale (**top row**), Mossdale (**middle row**), Whitendale (**bottom row**) for the burnt (Control; **left**) and mown (Treatment; **right**) sub-catchments. Different management started in April 2013 (pre versus post period). Also shown are pre- versus post-management means for each sub-catchment (i.e. far right line). Zero values are due to either snow or no flow and y-axis was truncated to 10 ppm. Radial and axial refer to the viewing of the plasma in the mass spectrometer analysis.

Overall, there were no significant differences in Zn, Cu, Pb and P concentrations between sites. However, Na, Mg, Mn, Al and K concentrations were significantly higher at Whitendale than at Mossdale or Nidderdale ($p < 0.001$ for all except for K were $p < 0.05$), Fe concentrations were significantly higher at Nidderdale ($p < 0.001$) and Ca concentrations were significantly higher at Mossdale ($p < 0.001$). Fe concentrations were greater under mowing at Whitendale ($p < 0.001$) but lower at Nidderdale and Mossdale, yet this was the case both pre-and post-management. Only P concentrations showed a noticeable and significant increase during the combined post-

management periods compared to the pre-management periods across all sites ($p = 0.090$), and a significant interaction between management and time period ($p< 0.01$), with P concentrations being 3 times higher in the post- compared to the pre-management period (i.e. as a ratio) for mown sub-catchments (ratio of 3.0±1.8) compared to burnt sub-catchments (ratio of 0.8±0.1) and without any difference in concentrations between catchments pre-management (mean of 0.03 ppm). This may indicate leaching from the brash, as indicated by the peak at Mossdale and Whitendale mown catchments in November 2013, the first year of management intervention, and less so again at Whitendale in September 2015 after the second mowing. This decline in a possible brash effect could relate to either a decline in overall management size (higher in 2013, i.e. 7 areas of an average 0.24 ha (total of 1.68 ha) in 2013 versus 5 areas of about 0.25 ha (total of 1.22 ha) in 2015) and/or proximity of mown areas to the central stream (closer in 2013) as shown in **Figure 2ab** (note colour coding for 1st and 2nd management intervention). Additionally, the large peaks in Fe (**Figure 63**) which were observed in summer coincided with peak summer stream flow volumes (**Table 10**) that were either side of a dry and warm period, potentially indicating that available Fe was being flushed out from both ash and brash layers. Notably, the biggest increase in Fe exports were observed at Nidderdale for the burnt and at Whitendale for the mown catchment indicating that losses can be large regardless of management, possibly depending on the amount of biomass (which was not determined for the entire managed areas).

The EMBER project (Brown et al., 2013) found elemental differences in stream water which were associated with lower pH and lower Ca concentrations, and which showed increased Al, Fe, Si and Mn in rivers draining burnt peatland rivers. Clay et al. (2010c) found Al and Fe concentrations increased in peatland soil solutions after burning, while Worrall & Adamson (2008) found a similar effect for Al only, indicating that there are some linkages between management that affect soil processes and the rivers draining those catchments. Whereas such changes could not be detected in this study, a noticeable increase in P concentrations in mown sub-catchments indicates some ecologically relevant implications of alternative management on nutrient loss at the ecosystem scale (although this would require some more detailed modelling study from stream to reservoir), where P probably leaches from the brash, which is potentially linked to monthly summer peak stream flow rates. Importantly, as only part of each sub-catchment was managed differently, implications of a complete management change and likely long-term trajectories for nutrient cycling and losses in stream water are unknown, and for P may include potential contribution to eutrophication issues for reservoirs. However, only K enrichment of surface peat was statistically different in the EMBER study and was attributed to ash and root decomposition (Brown et al., 2014). Although nutrients in the peat were not measured, this study observed a general increase in most nutrients in heather shoots on burnt and mown plots compared to uncut plots (see Section 4.2.5.1), but also only K was statistically different between burnt and mown plots. Noticeably, other nutrient concentrations and even pH in the surface peat were statistically unaffected by recent burns in the EMBER study (Brown et al., 2014), which is broadly in agreement with this study where there were no robust concentration changes over time in any of the elements in the flow samples apart from the increased P in mown sub-catchments.

4.2.11 *Fluvial carbon*

The long-term average monthly DOC and POC concentrations in stream flow across all sites and sub-catchments (25.8 mg L^{-1} and 2.2 mg L^{-1} respectively) were well within the range observed within the literature (Palmer et al., 2016) and agreed with a reported ratio of POC to DOC of 1:10 (Hope et al., 1997) for upland catchment rivers. The missing DOC carbon fraction caused by filtration was on average 8% (between the 0.70 µm filter used for POC and the 0.15 µm Rhizon sampler used for refiltering stream flow DOC samples) and 3% (between the 0.70 µm filter and the 0.45 µm filter frequently used for DOC analysis), compared to the amount of DOC in the filtrate obtained from the 0.70 µm filter used for the POC filtration (see Appendix 5 for further information).

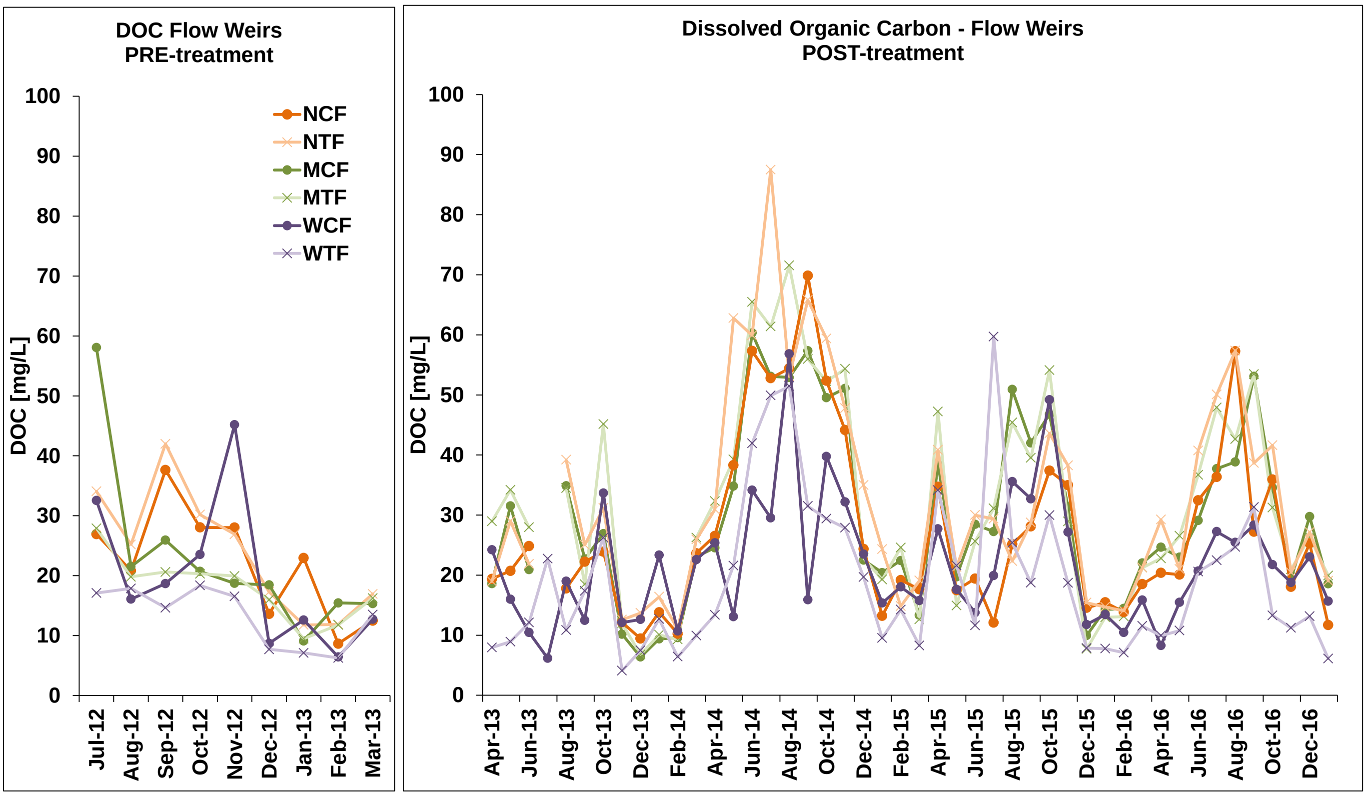


Fig. 64 Monthly stream flow dissolved organic carbon (DOC) concentrations (filtered by 0.15 µm Rhizon samplers) for the pre- (2012-2013**; left**) and post- (2013-2016**; right**) management periods for Nidderdale (N), Mossdale (M) and Whitendale (W) for the stream flow (F) samples from burnt control (C) and mown treatment (T) sub-catchments. Missing values are due to either snow or no flow.

The average DOC (0.15 µm) (**Figure 64**, above) and POC (**Figure 65**, below) concentrations were significantly ($p< 0.001$) different between sites, with DOC decreasing from Nidderdale (28.9 mg L^{-1}) to Mossdale (28.5 mg L^{-1}) to Whitendale (19.9 mg L^{-1}) and POC decreasing from Nidderdale (2.8 mg L^{-1}) to Whitendale (2.1 mg L^{-1}) to Mossdale (1.6 mg L^{-1}). However, the average modelled (i.e. upscaled by flow volume) monthly total export (calculated using rainfall and concentration) of DOC was highest at Mossdale (28.4 kg C ha^{-1}), lower at Nidderdale (22.4 kg C ha^{-1}) and lowest at Whitendale (20.9 kg C ha^{-1}). In contrast, the site differences for modelled monthly total POC export were similar to the measured concentrations with Nidderdale exporting the most (3.0 kg C ha^{-1}), Whitendale less (2.4 kg C ha^{-1}) and Mossdale the least (1.4 kg C ha^{-1}). Statistical analysis revealed strong site differences for all measures ($p< 0.001$) except for modelled monthly total DOC export ($p = 0.17$), but there was a significant impact of month for all DOC ($p< 0.05$) and POC ($p< 0.01$) estimates and of rainfall during the four weeks prior to sampling for DOC and POC concentrations (coefficient: -0.001; $p< 0.05$). Management interacted significantly with period (pre versus overall post) in explaining POC concentrations ($F_{1,\ 250} = 8.5$, $p< 0.01$) and there was a significant interaction between management, period and site for the modelled POC export totals ($F_{2,\ 147} = 7.2$, $p< 0.01$). This

likely related to the mown sub-catchment at Whitendale showing greater mean POC concentrations (2.7 mg L^{-1}) than the burnt sub-catchment (0.8 mg L^{-1}) during the entire post- but not the pre-management period. However, as a stream bank collapsed upstream of the flow weir in spring 2013, it is unlikely that this reflected a management impact, but rather a natural event. Moreover, at Nidderdale mean modelled POC export was significantly ($p < 0.05$) greater in the burnt (2.1 kg C ha^{-1}) than the mown (1.4 kg C ha^{-1}) catchment only during the entire post-period indicating a potential management impact (i.e. greater erosion from burnt areas).

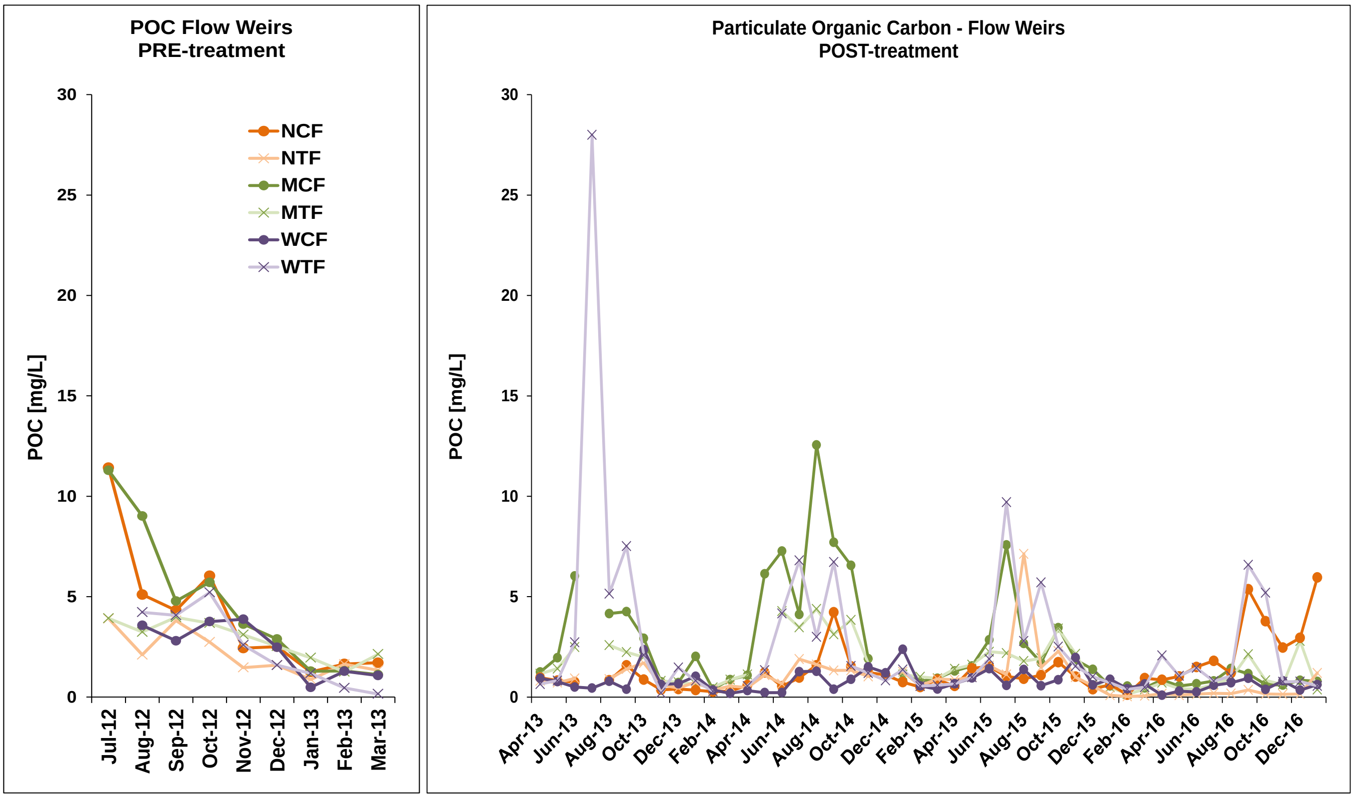
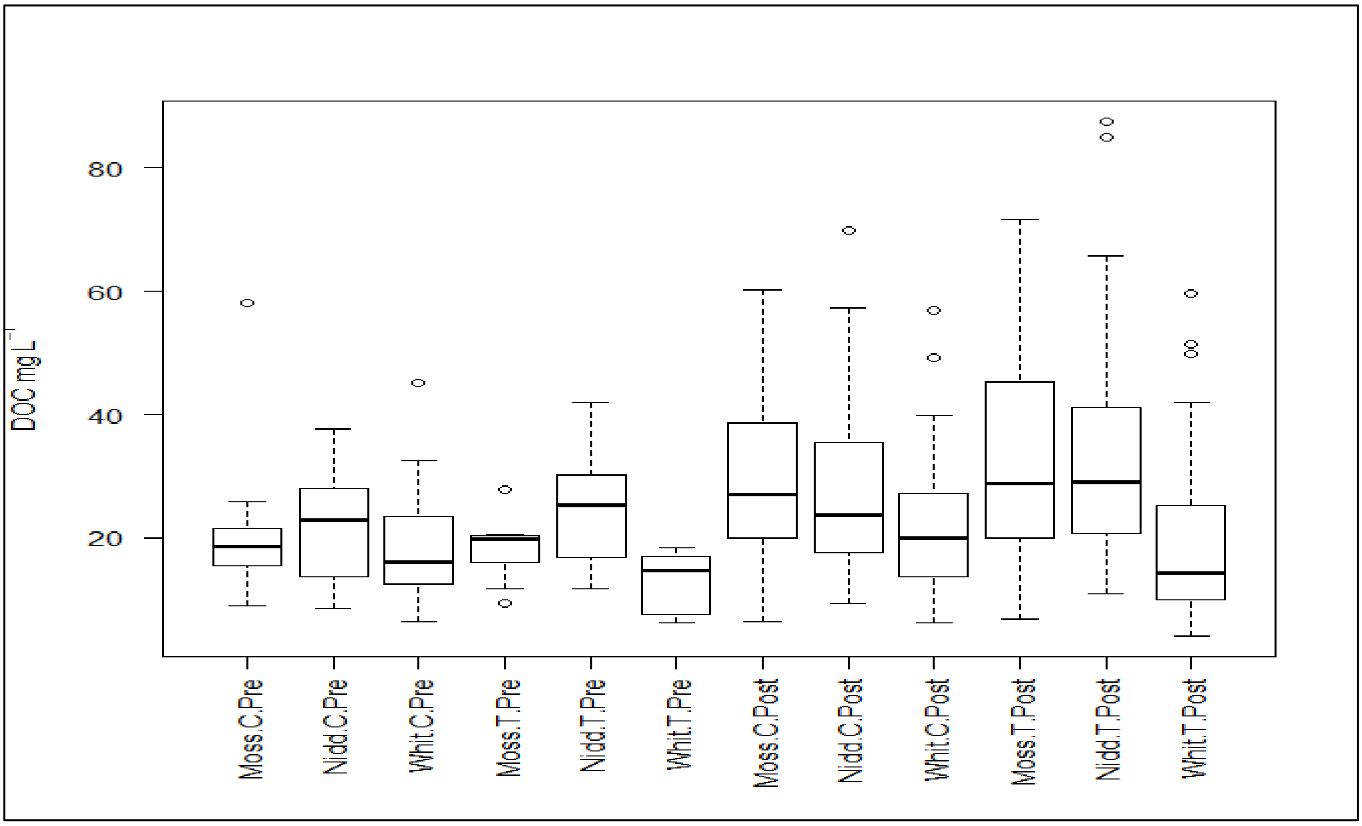


Fig. 65 Monthly stream flow particulate organic carbon (POC) concentrations for the pre- (2012-2013**; left**) and post- (2013-2016**; right**) management periods for Nidderdale (N), Mossdale (M) and Whitendale (W) for the stream flow (F) samples from burnt control (C) and mown treatment (T) sub-catchments. Missing values are due to either snow or no flow.

The average monthly DOC and POC concentrations during the pre- and post-management periods (**Figure 66** below) showed differences between sites and periods, but no consistent management effect. However, as to date only about 50% of the entire heather-dominated catchment area has been managed differently, changes in DOC and POC concentrations (and corresponding carbon export rates) in response to alternative management might only become apparent over a longer time scale and increased management intervention area.

Overall, the average annual post-management period DOC export rates (271-367 kg C ha^{-1} a^{-1}) were around 13 times higher than POC (17-36 kg C ha^{-1} a^{-1}) export rates (**Table 13**). Both values compared reasonably well to those reported by Hope et al. (1997), who measured 13-115 kg C ha^{-1} a^{-1} of DOC and 6-85 kg C ha^{-1} a^{-1} of POC for upland rivers. Notably, the sites used by Hope et al. (1997) included other soil types and larger catchments, and they found a strong positive relationship between peat areas on hill slopes and carbon export rates; here nearly only blanket bog was present and catchments were smaller (~1 km^2), which may explain the higher DOC export rates in our study as there was less opportunity for them to become diluted. During the post-management period, 2014, and particularly 2015, showed the highest DOC and POC export across all sites. These differences were positively related to annual rainfall amounts (**Table 1**) and flow rates (**Table 10**). As the total managed area was less in 2015 it is unlikely that this was management related (as this was not the case in 2013 with a larger managed area) but rather climate related. The differences in fluvial C export rates between sites reflect those observed for the modelled DOC and POC concentrations (see above), showing highest export rates for DOC yet

lowest for POC at Mossdale, with no significant overall management effect. However, a longer second post-management period would allow testing the increase in overall management area on stream C export rates but so far only one year after the second management phase was available limiting such an approach.



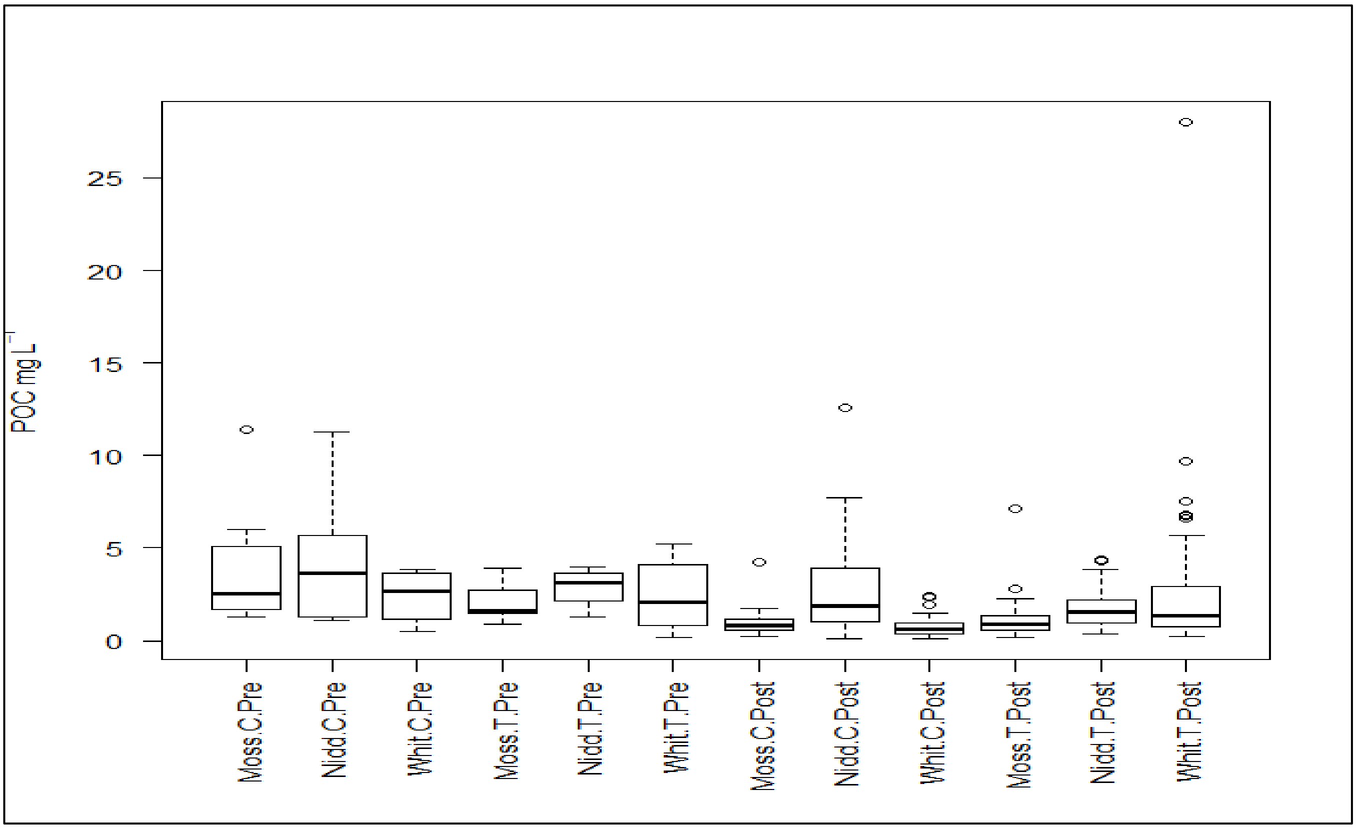


Fig. 66 Monthly DOC (**top**) and POC (**bottom**) concentrations (mg L^{-1}) for flow weir samples for Nidderdale (Nidd), Mossdale (Moss) and Whitendale (Whit) from the burnt control (C) and mown treatment (T) sub-catchments during the pre- (2012) and post-management (2013-2016) periods. The onset of different management was from April 2013. The box midline indicates the median, the box edges indicate the interquartile range and the points indicate data greater or less than 1.5 times the interquartile range.

Table 13 Calculated annual total dissolved organic carbon (DOC) and particulate organic carbon (POC) exports in stream water (from monthly flow samples) for Nidderdale (Nidd), Mossdale (Moss) and Whitendale (Whit) during the pre-management year 2012 (only from July; thus explaining higher values) and the post-management period (2013-2016). DOC_1 was derived by filtering water through Rhizon samplers (0.15 µm) also used for plot-level peat pore sampling and DOC_2 was the missing fraction (determined as 8%; see main text and Appendix 5 for the detailed methodology) of DOC in the filtrate compared to POC filtering (0.7 µm), providing an estimated total DOC_{1+2} (i.e. combined 0.15 and 0.7 µm filtration).

	C POC ($g\ m^{-2}$)		C DOC_1 ($g\ m^{-2}$)		C DOC_2 ($g\ m^{-2}$)				
	Pre-Burnt	**Pre-Mown**	**Pre-Burnt**	**Pre-Mown**	**Pre-Burnt**	**Pre-Mown**	**Annual estimate 2012**	**C POC ($g\ m^{-2}$)**	**C DOC_{1+2} ($g\ m^{-2}$)**
Nidd	10.5	8.7	36.6	53.8	2.9	4.3	Nidd	9.6	48.8
Moss	5.6	3.8	42.1	41.9	3.4	3.3	Moss	4.7	45.3
Whit	6.0	8.4	39.0	34.0	3.1	2.7	Whit	7.2	39.4
	Post-Burnt	**Post-Mown**	**Post-Burnt**	**Post-Mown**	**Post-Burnt**	**Post-Mown**	**Annual total 2013**	**C POC ($g\ m^{-2}$)**	**C DOC_{1+2} ($g\ m^{-2}$)**
Nidd	2.1	1.6	14.1	14.9	1.1	1.2	**Nidd**	1.8	15.6
Moss	0.7	0.7	21.5	20.6	1.7	1.7	**Moss**	0.7	22.7
Whit	0.8	2.8	13.1	10.3	1.1	0.8	**Whit**	1.8	12.7
	Burnt	**Mown**	**Burnt**	**Mown**	**Burnt**	**Mown**	**Annual total 2014**		
Nidd	3.3	1.5	24.8	31.7	2.0	2.5	**Nidd**	2.4	30.5
Moss	0.8	0.6	32.2	32.8	2.6	2.6	**Moss**	0.7	35.1
Whit	0.8	3.1	24.6	29.4	2.0	2.4	**Whit**	1.9	29.2
	Burnt	**Mown**	**Burnt**	**Mown**	**Burnt**	**Mown**	**Annual total 2015**		
Nidd	2.7	2.3	25.9	29.0	2.1	2.3	**Nidd**	2.5	29.6
Moss	1.1	1.7	44.3	39.2	3.5	3.1	**Moss**	1.4	45.1
Whit	1.6	3.8	27.0	28.9	2.2	2.3	**Whit**	2.7	30.2
	Burnt	**Mown**	**Burnt**	**Mown**	**Burnt**	**Mown**	**Annual total 2016**		
Nidd	1.8	1.1	20.3	17.9	1.6	1.4	**Nidd**	1.5	20.6
Moss	0.7	0.8	33.6	32.2	2.7	2.6	**Moss**	0.8	35.5
Whit	0.7	2.6	20.7	24.2	1.7	1.9	**Whit**	1.7	24.3

Interestingly, both the DOC and POC concentrations showed a weak positive linear relationship to the air temperature over the four weeks prior to taking monthly flow samples (concentrations were log transformed; R^2 = 0.16 and 0.18, respectively; **Figure 67**). Such a positive temperature response has been reported previously by Evans et al. (2016) in a global meta-analysis for DOC fluxes from near natural peatlands versus mean annual temperatures. The large amount of variability could be explained by differences in rainfall amounts causing concentration changes through stream dilution effects, as well as different areas of the catchments releasing different amounts of DOC and POC (as some areas so far are still not managed). Completing the full 10 year management cycle, more frequent sampling times or targeted sampling along recently managed plots (pore water filtrate and plot runoff) could thus reveal a much stronger relationship.

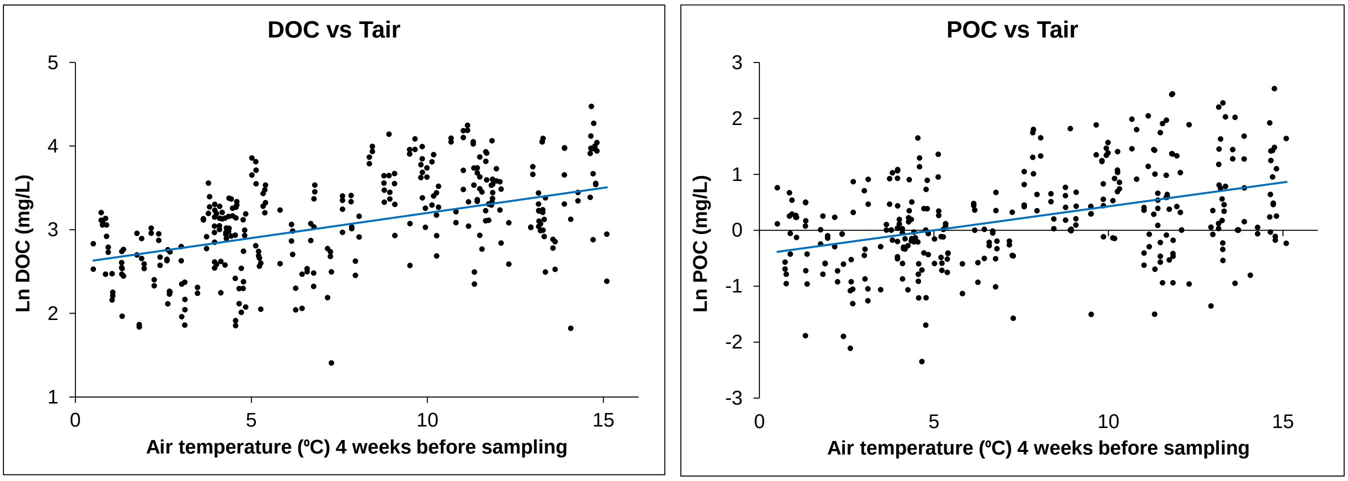


Fig. 67 DOC (**left**) and POC (**right**) log transformed concentrations (mg L^{-1}) for individual monthly flow samples across the three sites and their two sub-catchments versus the average air temperature during the four weeks before stream water samples were taken. Shown are the linear regression lines for the log transformed data (lnDOC = 0.063 * Tair + 2.60, adjusted R^2 = 0.16; lnPOC = 0.086 * Tair - 0.43, adjusted R^2 = 0.18).

In summary, the observations of plot pore water quality, stream water quality and fluvial carbon exports indicate that:

- Management intervention had no significant effect on pore water or stream water pH.
- Stream water and pore water pH increased across all management treatments and sites by about 1 unit over the study period, which was linked to soil temperature.
- Management intervention had no significant effect on pore water and stream water concentrations of DOC and colour index values.
- Pore water DOC and colour index values were higher in areas with sedge and *Sphagnum* cover and lower in areas with *Calluna* cover.
- Stream water phosphorus concentration increase was three times higher in the mown than burnt sub-catchments for post- versus pre-management periods.
- Overall C export as DOC was about 13 times higher than as POC.
- Management intervention did not cause any overall significant change in C export rates but highest export rates for DOC, yet lowest for POC, were observed at the wettest site.
- There was a positive effect of preceding air temperature on both pore water and stream water DOC concentrations, and on pore water colour index values.

4.2.12 *Soil temperature*

The EMBER study (Brown et al., 2015) reported a significant increase in surface peat temperatures after burning compared to mature heather stands. However, significant differences were only observed in the most recently burnt plots and likely reflected the observed very high maximum temperatures (more than 50°C). From an unpublished study (A. Heinemeyer, unpublished data), using the same sensors (Gemini PB-5001 thermistors interfaced with Gemini Tinytag TGP-4520 dataloggers), similarly high temperature maxima have been observed on blanket bog (see **Figures A1.4** and **A1.5** in Appendix 1). Importantly, this could be explained by parts of the long metal sensor being exposed to direct radiation, leading to artificially high temperatures being recorded, which is more likely when there is little vegetation cover. This can lead to short term sunshine related heat pulses and a bias when comparing mean temperatures in different age stands; notably, the box plots in Brown et al. (2015; cf. Fig. 3) show hardly any ecologically meaningful mean temperature differences between age stands when considering such potential bias. Therefore, care needs to be taken to prevent such artificial temperatures, which are not actually experienced by the peat surface. In this study, we repeated the same measurements using the same external sensors, ensuring that surface measurements were made with a moss or very shallow peat cover (0.5 cm) to prevent direct radiation warming any part of the metal sensor (**Figure 68**). For comparison, additional logger units (Gemini; Tinytag Plus 2; TGP 4017) with only an internal sensor and covered with a reflective shield were placed next to the external sensor units (**Figure 68**) across all plots during 2014-2015.

Fig. 68 A temperature logger with a reflective shield exposed during plot pore water sampling (**far left**) and a peat surface (under a thin 0.5 cm peat/moss layer) and a peat depth (under 5 cm of peat) soil temperature sensor (**left**). The two pictures on the **right** show the surface metal sensor before and after covering (yellow arrow) to prevent heating by the sun.

The comparison during 2014-2015 (i.e. during the post-management period) of surface temperatures obtained from covered internal sensors and protected external sensors revealed a very strong and, importantly, narrow relationship (**Figure 69**; left), with only a slight underestimation by the external sensor versus internal (across all burnt plots, the linear regression was: External temperature = 0.92*Internal temperature + 1.1; r^2 = 0.95). This comparison therefore confirmed that the covered sensors could be used to compare management impacts on surface temperatures during the entire monitoring period (i.e. when only internal sensors were used).

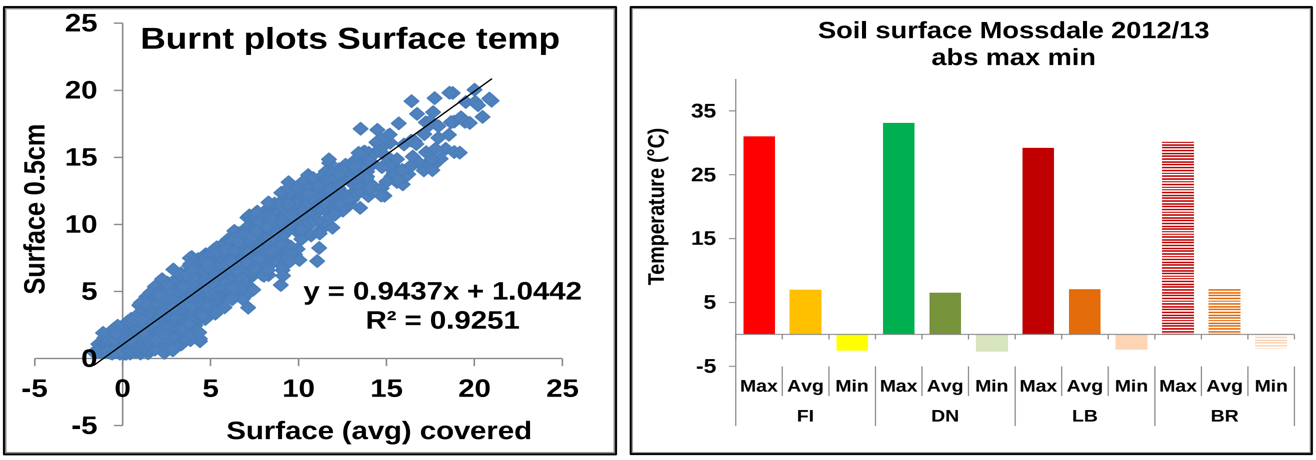


Fig. 69 Temperature comparison of external surface peat (0.5 cm) covered sensors and radiation shield covered internal sensor loggers from the burnt plots at Mossdale in 2014/15 (**left**) and the average, maximum and minimum temperatures from the surface loggers (**right**) during the pre-management period (2012/13) at Mossdale for the burnt (FI), uncut (DN), mown +brash (LB) and mown -brash (BR) plots. All plots had tall heather as this was before any management took place.

Across all sites, there were only small differences in the average, maximum and minimum surface temperatures between managements (see the Mossdale example in **Figures 69** and **70**). Before management took place (2012/13; **Figure 69**), some small differences in maximum temperatures were observed, but the average and minimum temperatures were very similar between plots. During the post-management period (2013 onwards; **Figure 70**), management affected maximum and minimum temperatures (but not the average), particularly on the mown (±brash) plots. The temperature amplitude on the mown plots with brash left (+brash) was generally less than on brash removal plots, indicating an insulation effect by the brash layer. Moreover, uncut plots showed the least negative minimum temperatures, reflecting protection by a closed canopy cover. Monthly summary tables for all three sites are provided in **Table A1.5abc** in Appendix 1.

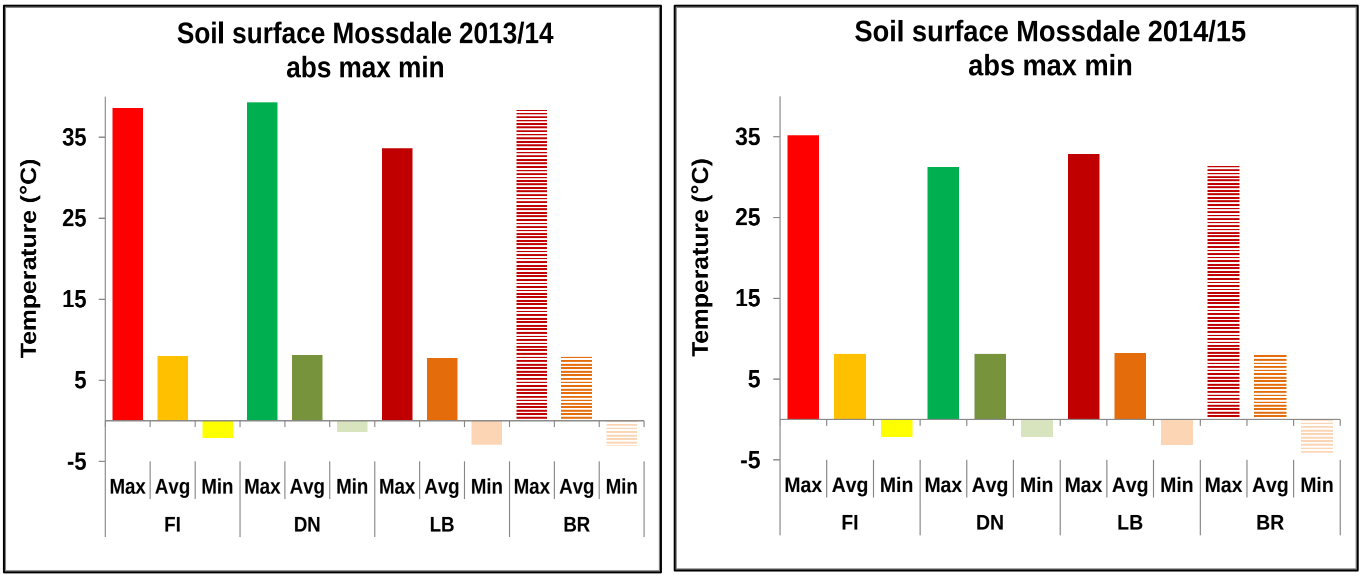


Fig. 70 Average, absolute maximum and minimum temperatures from surface loggers at Mossdale during the post-management period (April 2013-April 2014 (**left**) and April 2014-April 2015 (**right**)) for the burnt (FI), uncut (DN), mown +brash (LB) and mown -brash (BR) plots.

Most importantly, however, after management (from April 2013 onwards) burnt plots only showed a slight increase in the mean maximum surface temperatures (**Figure 70**) compared to the pre-management period (**Figure 69**; right), and no increase in the average temperatures (**Figure 70**). This highlights the relatively small impact of burning on actual peat surface temperatures compared to other factors affecting peat temperature, such as slope and aspect. For the three sites, the monthly temperature comparison between the main managements during the post-management period is summarised in **Table A1.5abc** (Appendix 1). For example, comparing burnt versus uncut during the year after management (April 2013 - March 2014; see **Figure 70** left for the Mossdale example), the average (absolute maximum/minimum) peat surface temperature averaged across all replicate sensors was: 7.6 (34.4/-3.6) versus 7.5 (36.9/-2.8) (Nidderdale); 8.0 (38.6/-2.2) vs 8.1 (39.3/-1.4) (Mossdale); 8.5 (39.9/-1.7) vs 8.2 (39.0/-0.9) (Whitendale). The same comparison for mown with left brash versus with brash removed was: 7.5 (34.3/-3.7) vs 7.6 (35.2/-4.2) (Nidderdale); 7.7 (33.6/-3.0) vs 7.9 (38.3/-3.0) (Mossdale); 8.4 (34.6/-1.9) vs 8.5 (33.6/-2.4) (Whitendale). However, immediately after burning, and particularly on bare and exposed peat, surface temperatures could increase to a greater extent, depending on wind and moisture conditions (Grau Andrés et al., 2017); although Grau Andrés et al. (2017) also showed not much of a change in average temperatures but a diurnal impact (i.e. both maxima and minima were affected but shifted in time). Monitoring plots in this study were randomly and permanently placed in vegetated and exposed areas reflecting the patchiness of vegetation and management through time, and therefore should reflect the overall impacts, averaged over the year after burning, across the management regimes.

4.2.13 *Soil respiration*

Soil respiration (SR) was measured ~4 times per year (see **Table A6.1** in Appendix 6 for dates) with a manual soil CO_2 flux chamber system (LiCor 8100) on vegetation-free areas (but leaving brash, litter or ash), which were maintained at each visit. These measurements were made on two sub-plots, either without or with root cutting, thus providing fluxes either including autotrophic root respiration (Ra) or fluxes excluding roots (SRc; i.e. roots were cut twice a year using a root cutter; **Figure 71**), respectively. From July 2015 onwards, total fluxes were also measured on net ecosystem exchange plots (NEEnew) within the re-growing areas where heather was cut or burnt in 2013. For these measurements, a 10 cm diameter area of moss was carefully removed within the NEE plot next to the larger plants (mostly heather and/or sedge) and replaced after the measurements, in order to prevent any change in soil moisture or root activity. Root respiration was calculated by subtracting fluxes of root free areas (SRc) from those including roots (SR or NEEnew), providing root respiration estimates (Rr or Ra NEEnew, respectively). Appendix 6 provides a more detailed summary of methods and data analysis.

Fig. 71 Soil respiration measurements with the Li-Cor 8100 CO_2 analyser using a soil chamber (10 cm diameter) on bare peat areas (**left**), which were either uncut or cut to 25 cm depth using a 15 cm diameter corer (**middle**), resulting in a clean cut surrounding the inner flux measurement area with a 2.5 cm outer buffer zone (**right**). Cutting was repeated twice a year.

Soil respiration (SR) fluxes on individual plots and measurement times ranged from 0.03 to 11.92 µmol CO_2 m^{-2} s^{-1} with a mean of 1.63 µmol CO_2 m^{-2} s^{-1}. The equivalent soil temperatures (Tsoil) ranged from -1.4°C to 35.8°C. There was no significant interaction between management and time period ($F_{5,\ 1208} = 1.81$, $p = 0.1088$), coinciding with the lack of differences in mean temperatures (**Figure 70**). The effect of site was marginally significant ($F_{2,\ 8} = 6.86$, $p = 0.019$); overall Mossdale had the highest SR and Whitendale the lowest (Mossdale: 1.94 ± 0.18 µmol CO_2 m^{-2} s^{-1}; Nidderdale: 1.59 ± 0.14 µmol CO_2 m^{-2} s^{-1}; Whitendale: 1.34 ± 0.13 µmol CO_2 m^{-2} s^{-1}), but there was no significant interaction between management, time period and site.

SR fluxes showed a significant positive non-linear relationship (**Table A6.4** in Appendix 6) with Tsoil ($Tsoil^2$: coefficient = 0.0008; $F_{1,\ 1092} = 146$, $p < 0.0001$). The month during which measurements were taken also significantly affected the SR flux ($F_{8,\ 940} = 65$, $p < 0.0001$), with greater fluxes occurring in summer months, particularly July. Similarly, the month of measurement significantly affected root respiration. This monthly effect was regardless of whether root respiration was measured on vegetation free (SRc; $F_{8,\ 112} = 12$, $p < 0.0001$) or within densely vegetated areas (Ra NEEnew; $F_{6,\ 413} = 23$, $p < 0.0001$), with the average root flux across all sites being 0.34 µmol CO_2 m^{-2} s^{-1} for Rr and 0.49 µmol CO_2 m^{-2} s^{-1} for Ra NEEnew. However, there were no Tsoil effects, nor any significant site or management differences, for root respiration (**Figure 72**; **Table A6.4** in Appendix 6). Decomposition fluxes obtained from cut, root-free areas (SRc) had an overall mean ± STDEV of 1.28 ± 1.29 µmol CO_2 m^{-2} s^{-1}. There was a strong Tsoil effect on SRc (i.e. soil decomposition) fluxes (coefficient = 0.029; $F_{1,\ 897} = 150$, $p < 0.001$; **Table A6.4** in Appendix 6) and a significant interaction between management and time period ($F_{3,\ 1216} = 3.9$, $p < 0.01$), with significantly lower decomposition fluxes on the mown plots in the post-management period where brash was removed (BR) compared to the mown plots with brash left (LB) and uncut (DN) plots (**Figure 72**). This probably reflected the higher amounts of decomposing litter.

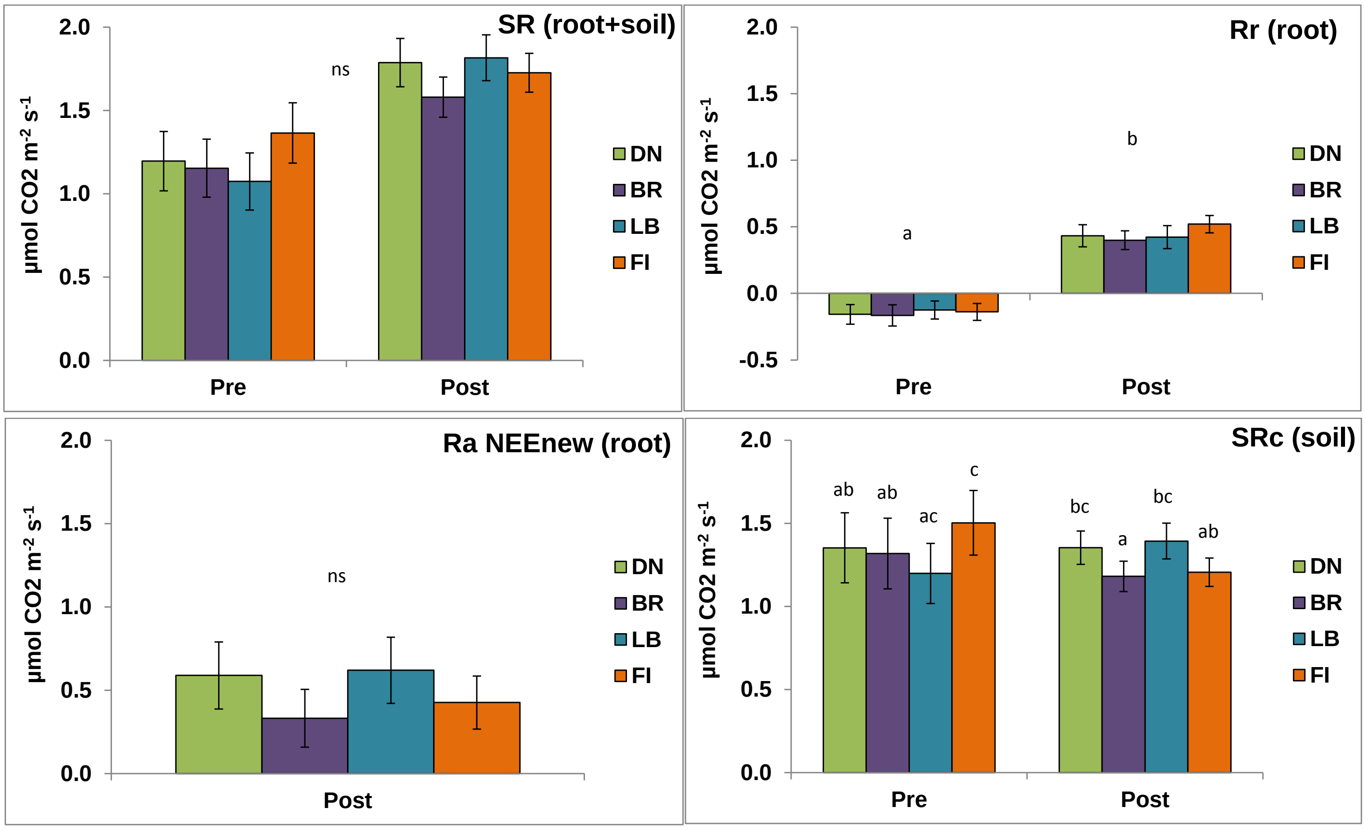


Fig. 72 Mean (±standard error) soil respiration (SR) fluxes for the total flux (**top left**), for only the root component (Rr) from vegetation free patches (**top right**), for only the root component from within vegetated soil patches (Ra) (**bottom left**) and for the soil decomposition (SRc) component (**bottom right**). All fluxes are averaged across the three sites and are shown for the pre- (2012) and post- (2013-2016) management periods, except the root fluxes from vegetated areas, which were only measured in 2016. Note that negative fluxes for root fluxes are a result of initial cutting causing enhanced root decomposition fluxes. Management codes (with combined *Sphagnum* treatments on mown plots) are: DN = uncut; BR = mown with brash removed; LB = mown left brash; FI = burnt. Significant differences are indicated with different letters with ns indicating no significant differences between treatments.

Root flux contributions (Rr) between treatments were not statistically different averaging around 26% of the total soil flux (**Figure 73**), about half that observed by Heinemeyer et al. (2011) on a blanket bog at Moor House NNR.

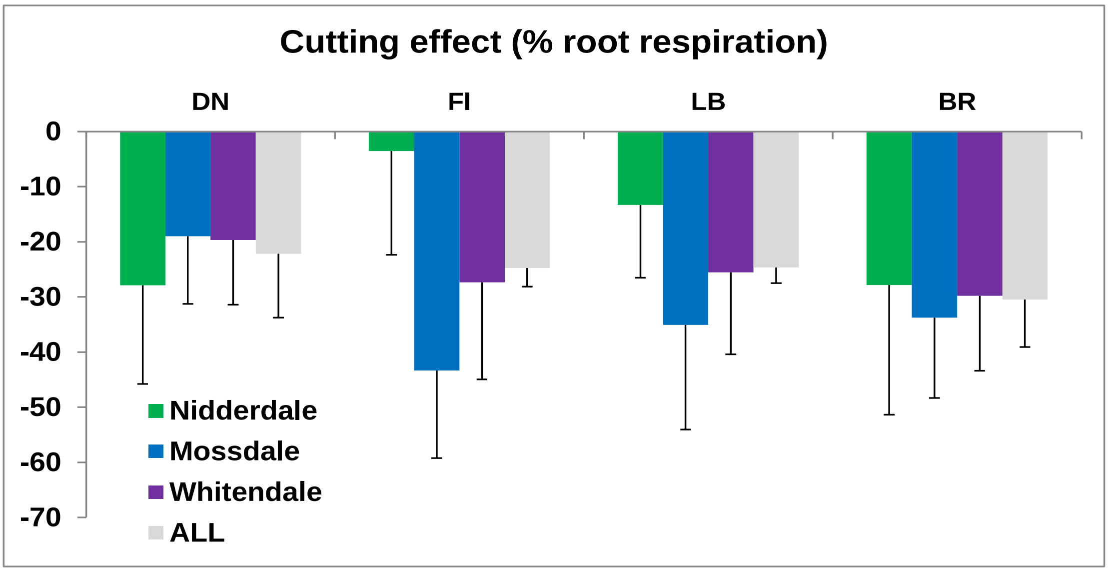


Fig. 73 Estimated amount of root flux (Rr) contribution (in percent - standard error) of the total soil flux as determined during post-management period (2013-2014) by the amount of the lost flux due to cutting of roots in vegetation free peat areas at the individual sites (Nidderdale, Mossdale, Whitendale). Management is indicated as uncut (DN), burnt (FI), mown with left brash (LB) or with brash removed (BR) both with combined *Sphagnum* pellet additions.

Importantly, the Heinemeyer et al. (2011) study used permanent deep collars to cut off roots, causing considerable water logging inside (as discussed in Heinemeyer et al. 2011); this reduced decomposition fluxes, and thus suggested an artificially high estimate of the percentage root flux contribution. The present study avoided this artefact and thus provides a more reliable estimate of the contribution of root fluxes. These seem to be the only available flux component data for UK peatlands showing decomposition versus root-derived components.

Average soil decomposition fluxes (SRc) were very similar between pre- and post-management periods, although post-management fluxes were slightly lower, despite higher mean soil temperatures (**Table 14**; left). Calculated temperature sensitivities (Q_{10}) values (see Heinemeyer et al., 2012 and Appendix 6 for the method) were also assessed. Such Q_{10} values are an indication on how decomposition (measured as CO_2 flux) increases in relation to a 10°C rise in temperature. For example a Q_{10} of 2 implies a doubling of flux for a temperature rise by 10°C. The observed Q_{10} values (**Table 14**; right) were well within the reported region of 2-4 for peatland soil CO_2 fluxes at Moor House (Briones et al., 2004). However, the post-management period consistently showed overall higher Q_{10} values than the pre-management period at all sites (**Table 14**; right). Although the SRc fluxes (i.e. cut decomposition only fluxes) showed slightly higher Q_{10} values than the uncut SR (i.e. total flux), this was unlikely to be significant (based on the overlapping standard error; SE), Q_{10} values were always substantially lower for the Ra NEEnew fluxes on the vegetated areas (i.e. with more roots). This agreed with previous findings for forests that the autotrophic root-derived flux (including the mycorrhizal component) is less temperature-sensitive than the decomposition component (Heinemeyer et al., 2007), highlighting the importance of including a root-free treatment to assess soil C impacts.

Table 14 Field soil respiration CO_2 fluxes (mean flux) for pre- (2012) and post-management (2013-2016) periods. **Left**: average soil temperature (Tsoil) and decomposition only soil fluxes (SRc) for pre- (n = 3) and post-management (n = 15) periods for each site and management. Plot replications were n = 4 (for uncut and burnt) and n = 8 for mown plots with *Sphagnum* additions combined (left brash, LB and brash removed, BR). **Right**: Flux responses to soil temperature (Q_{10} ± standard error; SE) for each management (DN = uncut; FI = burnt; LB = mown left brash; BR = mown brash removed) and collar treatment (SR = with roots; SRc = cut roots) combined across all three sites. NEEnew indicates soil respiration measurements within densely vegetated areas (i.e. more roots present then in the vegetation free areas).

Pre management	Mean Tsoil (°C)	Mean Flux ($\mu mol\ m^{-2}\ s^{-1}$)	Post management	Mean Tsoil (°C)	Mean Flux ($\mu mol\ m^{-2}\ s^{-1}$)
Nidderdale					
DN	12.7	1.58	**DN**	14.7	1.41
FI	12.6	1.58	**FI**	14.1	1.30
LB	12.7	1.46	**LB**	14.6	1.44
BR	12.6	1.49	**BR**	14.6	1.14
Mossdale					
DN	9.5	1.57	**DN**	15.2	1.46
FI	8.0	1.72	**FI**	15.2	1.40
LB	9.5	1.30	**LB**	15.3	1.54
BR	9.5	1.59	**BR**	15.3	1.32
Whitendale					
DN	7.8	0.90	**DN**	14.5	1.19
FI	8.0	1.21	**FI**	12.8	0.92
LB	7.7	0.83	**LB**	14.5	1.20
BR	7.7	0.87	**BR**	14.6	1.09

Q10s	Pre	SE	Post	SE
DN-SR	2.69	0.27	3.15	0.23
DN-SRc	2.80	0.32	2.98	0.23
DN-NEEnew	na	na	2.08	0.25
FI-SR	2.40	0.26	3.06	0.17
FI-SRc	2.49	0.26	3.35	0.21
FI-NEEnew	na	na	2.22	0.22
LB-SR	2.70	0.19	3.23	0.14
LB-SRc	2.72	0.21	3.25	0.15
LB-NEEnew	na	na	2.69	0.22
BR-SR	2.43	0.19	3.34	0.15
BR-SRc	2.68	0.21	3.39	0.16
BR-NEEnew	na	na	2.45	0.20

Importantly, there was no indication that burning enhanced microbial decomposition, as both the Q_{10} values (**Table 14**) and the overall mean fluxes (**Figure 72**) did not differ significantly between burnt plots and any other management treatment during the post-management period. In fact, mean fluxes were lowest for burnt (FI) and brash removal (BR) plots at all sites post-management (**Table 14**), whilst both the highest fluxes and high Q_{10} values were observed for the left brash (LB) mown plots, indicating considerable decomposition from the brash.

However, as recent management mainly affected the top layer of the peat (i.e. either leaving brash or adding ash and charcoal), the lack of significant differences in the field study may reflect a large (albeit as yet unknown) proportion of the soil flux from the peat below the surface layer. Therefore, additional laboratory investigations were carried out to investigate effects on the surface 20 cm of the peat profile, and specifically the top 5 cm in relation to deeper peat layers, to analyse chemical composition and decomposition products under the different treatments, and to investigate the effects of charcoal and mycorrhizal priming on decomposition rates. These experiments are described in Sections 4.4.2, 4.4.3. and 4.4.4.

In summary, the field soil chamber flux measurements showed that:

- There was little field evidence that management intervention significantly affected soil respiration fluxes, the absolute rate of decomposition, or its Q_{10} value.
- However, these field measurements included CO_2 fluxes from throughout the peat layer. Management intervention, in contrast, would be most likely to affect decomposition processes close to the surface of the peat.

4.2.14 *Net ecosystem exchange*

Net ecosystem exchange (NEE) was measured 4 times per year using a Li-Cor 8100 CO_2 flux analyser connected to a custom made Perspex chamber of adaptable size to reflect vegetation height differences and re-growth (**Figure 74**). Crucially, in peatlands (in fact, in most ecosystems) fine root distribution is largely concentrated in the top 5 cm of soil (see Heinemeyer et al., 2011). Therefore, no permanent collar was inserted as this would have potentially damaged plants, reduced root fluxes and altered peat drainage and decomposition fluxes (see Heinemeyer et al., 2011). Instead, a wet moss 'roll' was gently pressed against the outside of the chamber (see **Figure 74**) to provide an airtight seal, which was removed after each measurement. Moreover, to allow light response curves to be derived for later use in upscaling measured fluxes over time under varying light conditions, measurements were made in full light, under shade (mesh) and in the dark (hood), whilst light and temperature conditions were monitored within the chamber (see **Figure 74**). A pressure vent (Li-Cor) was used to avoid over-pressuring during chamber placement, whilst also limiting Venturi (wind) effects. Appendix 6 provides more detailed methodological information and a summary of methods of data analysis.

Fig. 74 Net ecosystem exchange measurements with the Li-Cor 8100 (**left**) using a custom made 30 cm diameter Perspex chamber of varying height (adjusted to vegetation level – maximum of 60 cm; **2nd from right**). Measurements were done either during full light, shading (single: 70% shade; double: 90% shade) or in the dark (ecosystem respiration) to obtain light response curves (**3 central pictures**). Temperature (shielded from the sun) and light (PAR) sensors were fitted within the chamber and two fans aided air mixing in larger chambers (**far right**). Chambers were not placed on collars inserted into the peat, but instead used a ring of removable *Sphagnum* moss to seal the chamber base around the peat surface.

Heather plants can reach substantial size with a considerable shoot biomass, possibly taking up a considerable part of the chamber volume. As NEE flux calculations are based on the chamber volume, it could be important to correct for plant sizes affecting the actual internal chamber gas volume. Therefore, plant volumes were estimated in the field and these estimates were validated in the laboratory (see previous Section 4.2.5.2 for further details).

Based on using the tallest NEE chamber (height 60 cm; volume of 39.6 L), the difference between NEE fluxes corrected for *Calluna* volume (see **Figure 28** in Section 4.2.5.2) and uncorrected fluxes was significantly different (paired Student's t-test: $t_{791} = 10.00$, $p< 0.0001$) and ranged from -0.59 to 0.76 µmol CO_2 m^{-2} s^{-1} (mean of 0.04 µmol CO_2 m^{-2} s^{-1}). However, flux changes for plant volumes occupying less than 2.5% of the chamber volume of c. 40 L (= <1 L) were within the normal variation (measurement noise) of light and dark flux measurements (**Figure 75**). The results indicated that larger plant volumes should be incorporated into NEE flux calculations (Morton & Heinemeyer, 2018). The larger the NEE fluxes and the larger the plants are in relation to the chamber, the greater the absolute change in NEE fluxes. This was of particular importance when NEE fluxes were up-scaled, as any errors in flux calculations would be multiplied. However, mowing and burning removed the majority of the *Calluna* biomass from the plots and post-management most plants were estimated to take up less than 1.5% of the smaller chamber volume (height 20 cm; volume of 12.3 L), which had no significant effect on flux calculations. Consequently, corrections were only applied for the uncut (DN) plots post-management.

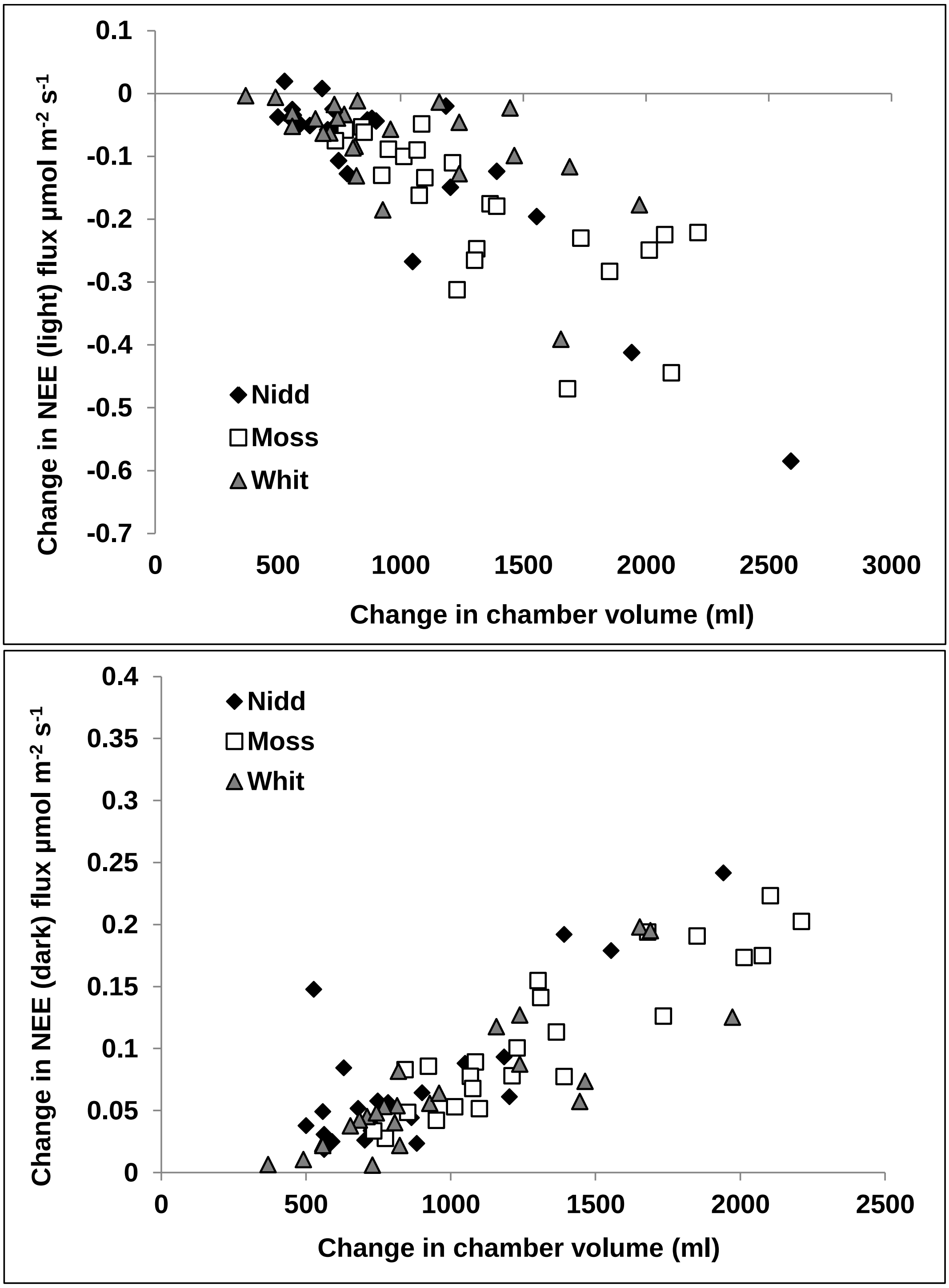


Fig. 75 Predicted changes in net ecosystem exchange (NEE) fluxes in October 2012 for all plots at the three sites (Nidderdale = Nidd; Mossdale = Moss; Whitendale = Whit) based on the *Calluna* plant volume (see **Figure 28**) decreasing the effective chamber air volume used in flux calculations in the light (**top**) and dark (**bottom**) within the NEE flux chambers (diameter of 30 cm with a height of 60 cm and a volume of 39.6 L). Mean NEE fluxes (±SE) for the three sites (Nidd, Moss and Whit) were: -3.70±0.59, -5.39±0.49, -3.11±0.49 µmol CO_2 m^{-2} s^{-1} (light); 3.40±0.45, 3.18± 0.20, 2.52±0.25 µmol CO_2 m^{-2} s^{-1} (dark).

The volume corrected Full Light NEE CO_2 flux measurements (i.e. negative values represent a CO_2 uptake) ranged from -19.91 to 9.02 µmol CO_2 m^{-2} s^{-1}, with a mean of -0.89 µmol CO_2 m^{-2} s^{-1}, while the corrected dark ecosystem respiration R_{eco} fluxes were between 0.03 and 27.73 µmol CO_2 m^{-2} s^{-1} (mean of 3.00 µmol CO_2 m^{-2} s^{-1}). Shaded fluxes were about 30% of the light measurements (reflecting the percentage in light reduction) and were used for upscaling. During Full Light measurements, the photosynthetically active radiation (PAR) ranged from 12 to 1951 µmol m^{-2} s^{-1} (mean of 547 µmol m^{-2} s^{-1}). Soil temperatures were very similar between pairs of the Full Light and R_{eco} measurements (over 97% varied by less than 2°C), but varied across all measurements from -3°C to +37°C.

There was a significant interaction between management and time period for both Full Light and R_{eco} NEE fluxes (**Figure 76**) ($F_{5,\ 1210}$ = 7.58, $p< 0.0001$ and $F_{5,\ 1195}$ = 15.22, $p< 0.0001$, respectively). There were no significant differences between management treatments in 2012 under either light condition. Post-management, as expected uncut DN plots took up significantly more CO_2 than any other management under Full Light conditions $p< 0.001$ for all) and emitted significantly more CO_2 during R_{eco} fluxes ($p< 0.001$ for all). Across the post-management period, FI plots lost CO_2 in Full Light, resulting in significantly lower fluxes than both mown (LB and BR) managements ($p< 0.02$) showing a small uptake in full light. Conversely, both mown treatments had higher dark R_{eco} CO_2 fluxes than FI plots ($p< 0.01$).

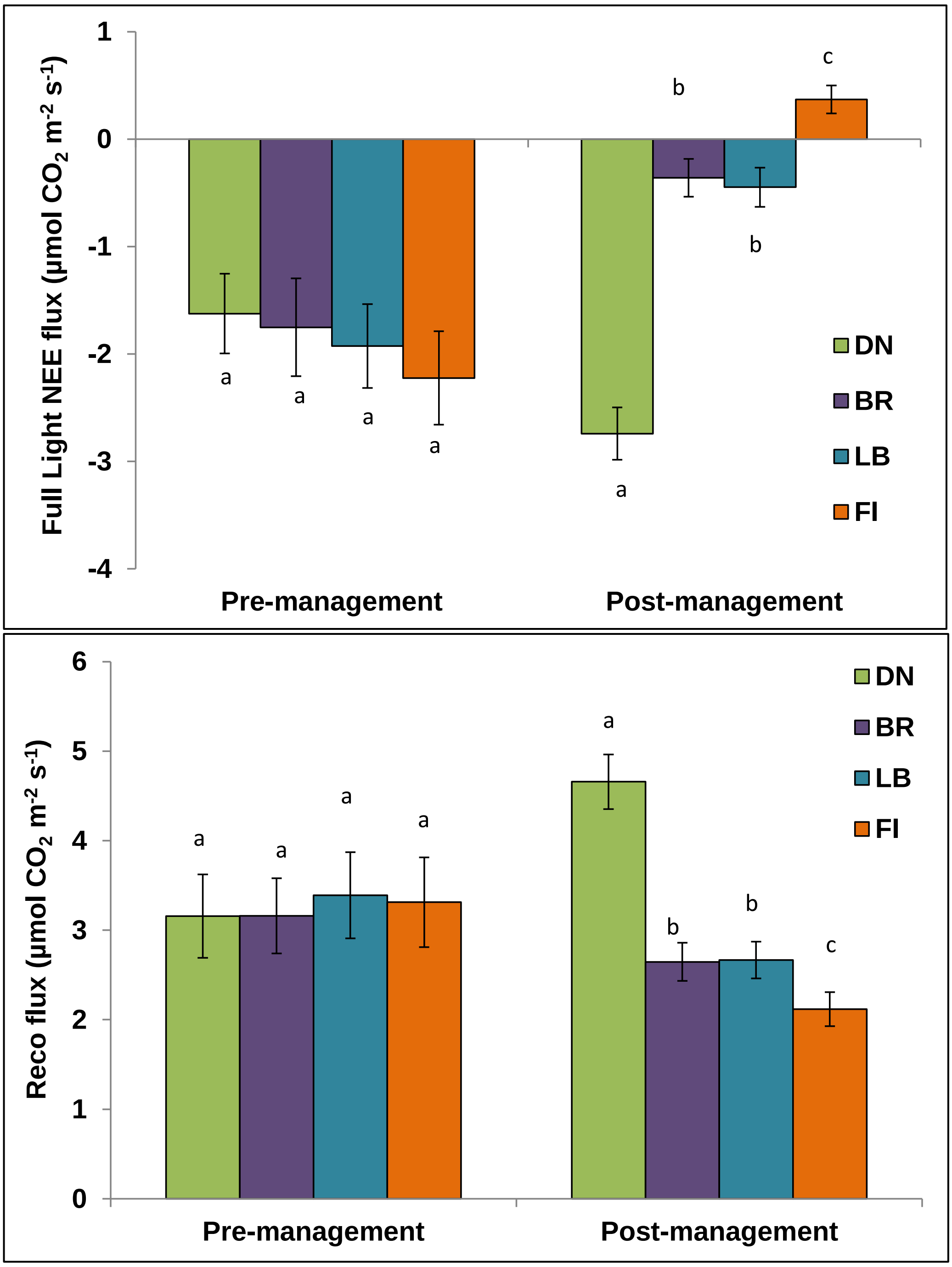


Fig. 76 Mean (± standard error) of the Full Light (**top**) and R_{eco} (**bottom**) components of the net ecosystem exchange (NEE) fluxes during the pre- (2012) and post- (2013-2016) management periods for each management averaged across all three sites (DN = uncut; BR and LB = mown with either brash removed or left, respectively, both with combined *Sphagnum* treatment; FI = burnt). Different letters within each time period indicate significant differences between managements per period.

Both Full Light and dark R_{eco} NEE fluxes differed significantly between sites ($F_{2,7} = 5.80$, $p = 0.0357$ and $F_{2,6} = 8.56$, $p = 0.0158$, respectively), with Mossdale having the highest CO_2 uptake in Full Light conditions (**Figure 77)** and greatest CO_2 loss during R_{eco} measurements (**Figure 78**). There were no significant differences in either of the parameters at any of the sites before management intervention. There was a three-way interaction between management, time period and site for Full Light NEE fluxes ($F_{10,1195} = 2.14$, $p = 0.0192$) and a marginally significant interaction for R_{eco} NEE fluxes ($F_{10,1195} = 1.76$, $p = 0.064$). As expected, during post-management at all sites uncut (DN) plots showed the largest CO_2 uptake in Full Light conditions and also the highest dark R_{eco} fluxes ($p < 0.001$ for all). Whilst at Nidderdale the other plots did not show differences between managements, at Mossdale, burnt (FI) plots showed lower R_{eco} fluxes than mown LB ($p < 0.02$), and at Whitendale FI plots had lower uptake in full light ($p < 0.001$) and also emitted significantly less CO_2 during R_{eco} than LB and BR plots ($p < 0.01$ for both).

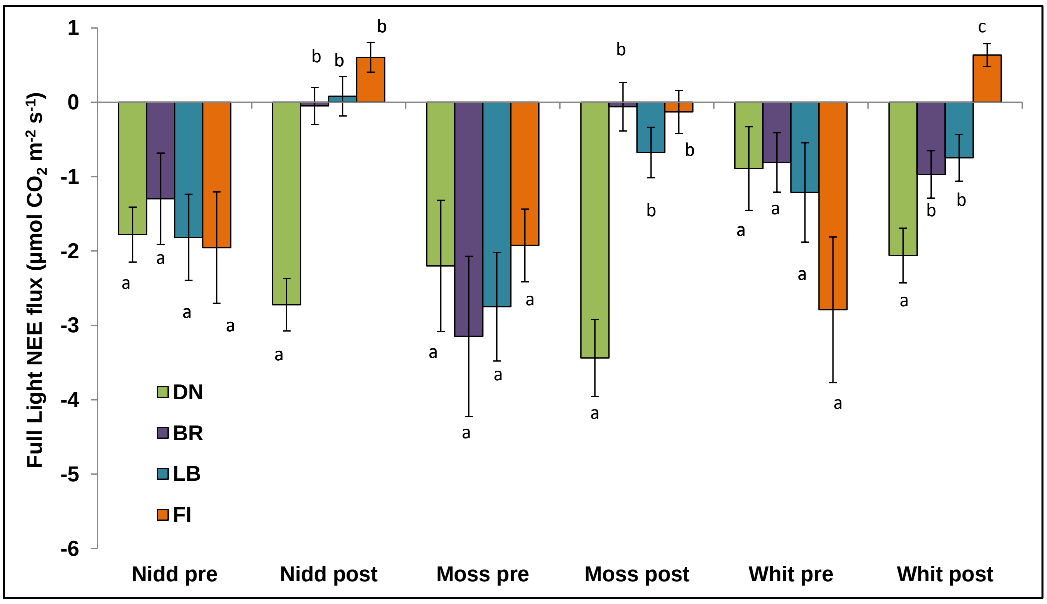


Fig. 77 Means (± standard error) of Full Light net ecosystem exchange (NEE) fluxes of each site (Nidderdale, Nidd; Mossdale, Moss; Whitendale, Whit) for the pre- (2012) and post-management (2013-2016) periods. Management codes were DN (uncut), FI (burnt), BR & LB = mown with either brash removed or left, respectively, both with combined *Sphagnum* treatments. Different letters within each time period indicate significant differences between managements.

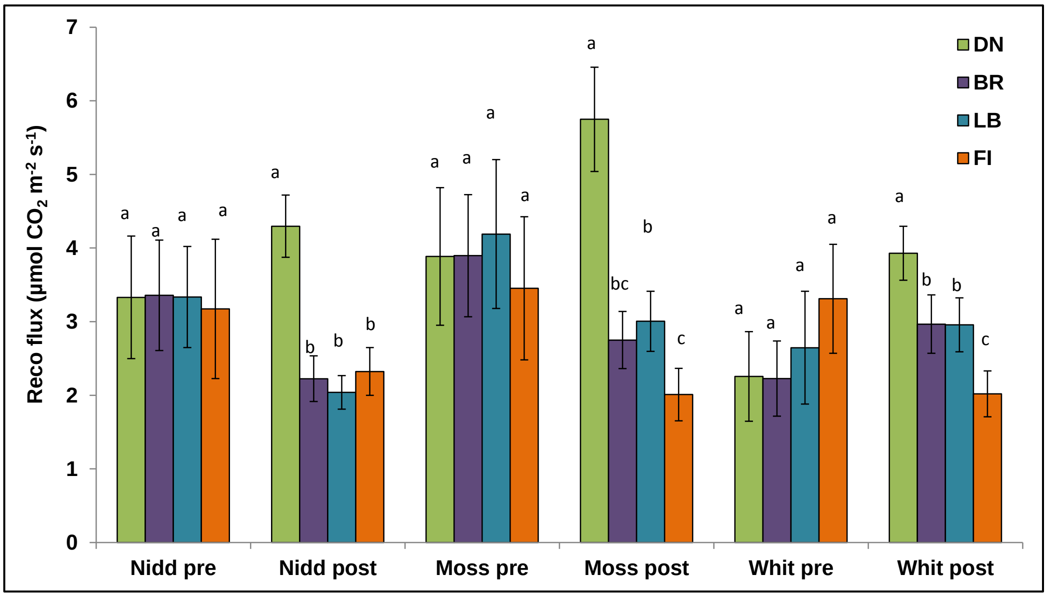


Fig. 78 Means (± standard error) of ecosystem respiration (R_{eco}) fluxes from dark chamber fluxes for each site (Nidderdale, Nidd; Mossdale, Moss; Whitendale, Whit)during the pre- (2012) and post-management (2013-2016) periods. Management codes are as in **Figure 77**. Different letters within each time period indicate significant differences between managements (although this was only marginally significant with a p-value of 0.064).

Environmental controls on both flux components were included in an overall statistical analysis. PAR had a non-linear effect on the Full Light fluxes (**Table A6.4** in Appendix 6); a negative PAR coefficient and the smaller positive PAR^2 coefficient together demonstrate that an incremental increase in PAR increased CO_2 uptake but by a smaller amount than the previous increment. Likewise, soil temperature (Tsoil) had a similar non-linear effect on R_{eco} fluxes (**Table A6.4**); the linear Tsoil coefficient was positive and the $Tsoil^2$ coefficient was negative, demonstrating that with every unit increase in temperature, the increase in CO_2 flux declined (over the measured range). Soil temperature also significantly affected Full Light NEE fluxes (**Table A6.4**), although the overall direction of the effect was unclear due to increasing temperature causing both greater uptake and release of CO_2. Despite the inclusion of both PAR and soil temperature in the analysis, the month in which measurements were taken also significantly affected both the Full Light NEE and R_{eco} fluxes (**Table A6.4**), suggesting that other seasonal changes, including general (i.e. monthly) light levels and phenology were important in addition to the *in situ* light levels and soil temperatures during flux measurements.

Discussion

The management types which showed high CO_2 uptake during Full Light flux measurements also showed high CO_2 release during R_{eco} flux measurements (**Figures 77** and **78**). This effect is likely due to the size of the plants, particularly as the uncut (DN) plots showed both the greatest drawdown and release of CO_2 in full light and dark (R_{eco}) conditions, respectively. Larger plants usually have more leaves, and hence should have a greater photosynthetic capacity, which would explain the greater CO_2 uptake during light conditions. Larger plants also are likely to produce more respiration, not only from their leaves and roots, but also from decomposition of leaf and root litter, which is likely to be greater beneath larger plants (Brown & MacFadyen, 1969). Dixon et al. (2015) demonstrated that NEE fluxes were best explained by the height of the heather plants on blanket bogs dominated by heather in the South Pennines and Peak District, which would support this suggestion.

However, Dixon et al. (2015) concluded that, where heather was dominant, the NEE flux would almost always result in a net loss of CO_2 and that this loss would increase with heather height. In the present study, the difference between the average post-management uncut (DN) Full Light and R_{eco} fluxes was 1.92 µmol CO_2 m^{-2} s^{-1}, which was smaller than the difference between the equivalent averages for any other management (compare **Figures 77** and **78**). In our study, the other post-management flux differences for the regrowing plants were: for FI 2.5 µmol CO_2 m^{-2} s^{-1}, for LB 2.2 µmol CO_2 m^{-2} $s^{-1}$and for BR 2.3 µmol CO_2 m^{-2} s^{-1}. This suggests that the net CO_2 loss does not necessarily increase when taller heather plants are present. Clearly, extending the monitoring period would give flux values over substantially more of the life cycle of heather, which would address an important knowledge gap in relation to age class carbon sequestration potential of heather-dominated blanket bog. This evidence could then be used to derive recommended optimum rotation length in view of maximising C sequestration. The difference between the present study and that of Dixon et al. (2015) could also lie in the area studied: their study was conducted in the South Pennines and Peak District which have become virtually devoid of *Sphagnum* mosses due to historic air pollution, whereas all sites in this study possessed some *Sphagnum*, as well as *Eriophorum*, which would have contributed to carbon drawdown by photosynthesis. Alternatively, or additionally, Dixon et al. (2015) inserted their collars into the peat by up to 5 cm, severing surface roots and thus enhancing decomposition and altering drainage (Heinemeyer et al., 2011), whereas collars in the present study were only placed on the surface.

It is unlikely that a difference in temperature caused the differences in NEE fluxes between sites as the difference in annual average temperature between the three sites was only 0.5°C (**Table 1**). It is also unlikely that site differences were caused by different light levels occurring during measurements, as Nidderdale received most light on average (**Table 1**) and took up least CO_2 during Full Light measurements. Plant size is a more likely contributing factor to the site differences as Mossdale had both the largest plants (greatest volume; **Figure 28**)

and the highest CO_2 uptake and loss for Full Light and R_{eco} fluxes, respectively. The substantially higher cover of *Sphagnum* mosses at Mossdale compared to the other two sites (see **Figure 38**) may have also contributed to Mossdale having the highest (i.e. most negative) Full Light NEE fluxes, as *Sphagnum*-dominated peat has been shown to release less CO_2 than peat dominated by other species (Dunn et al., 2015). Whilst high *Sphagnum* cover does not explain why Mossdale released most CO_2 in the dark, the greater size of the heather plants (and thus biomass respiration) may have offset the presence of *Sphagnum*. Differences in management techniques between the sites may also have accounted for some of the measured difference in the fluxes.

Importantly, the absence of significant differences in NEE fluxes before management implementation gives credence to the post-management differences occurring as a direct consequence of the managements. Perhaps of greatest note is that plots under burn (FI) management were the only areas to consistently lose CO_2 during Full Light NEE measurements across the five-year post-management period (**Figure 77**). However, the dark R_{eco} fluxes (i.e. respiratory C losses) were also lower, which may be explained by the lower peat decomposition fluxes (SRc) on FI plots (see Section 4.2.13), possibly due to charcoal impacts (see Section 4.4.2 and 4.4.4). Therefore the net carbon balance in the long-term might be more C sequestration. This is consistent with the greater peat accumulation (see Section 4.3.3) on the more frequently burnt sites (Nidderdale and Whitendale), although we acknowledge the limitations due to uncertainties in the dating of the peat cohort layers. Notably the flux method does include emissions from very old peat, possibly affecting or even obscuring the signal from the peat surface (which was assessed in Section 4.4.2), whereas a stock inventory allows looking for changes over particular time scales, but mostly depends on uncertain dating techniques (which are particularly less robust over more recent periods).

Up-scaling

The main aim of obtaining manual chamber fluxes for NEE, R_{eco} and SR flux components was to allow upscaling the net carbon flux balance over time based on NEE light response curves. This approach used regressions of NEE fluxes over a range of light conditions, which provided R_{eco} and maximum NEE (NEEmax) values, together with relationships of these parameters to environmental factors such as temperature and light. Commonly, eddy covariance (EC) flux towers are used to provide a continuous flux record, which then allows calculation of the net annual flux and its main components. Although sites in this project did not include EC flux towers, the time-course of similar plot-level manual chamber fluxes correlated well with that of EC data on a blanket bog in Wales near Lake Vyrnwy (A. Heinemeyer, unpublished data; **Figure 79**). Moreover, manual chambers also offer the advantage of combining NEE measurements with soil chamber flux components to allow the total R_{eco} flux to be broken down into its components (i.e. respiration from shoots, roots and soil decomposition), which is not possible with the EC method on its own (Heinemeyer et al., 2012).

Crucially, only the inclusion of a chamber approach allows an estimation of net primary productivity and carbon use efficiency values, both vital but rarely measured ecosystem parameters. This reflects the ability of ground chambers to provide the necessary soil flux components of root versus decomposition (i.e. to separate R_{eco} into plant respiration and soil decomposition). However, hardly any comparisons of these two methods exist for NEE fluxes on peatlands. Previous work on *Eriophorum*-dominated tundra by Oechel et al. (1998) showed good agreement across scales, although the authors also highlighted the issues of heterogeneous systems, as did Fox et al. (2008). However, the blanket bogs in this study were very uniform in structure. In fact, a comparison of corresponding daily NEE predictions for Mossdale to EC flux tower data from Vyrnwy, which had a very similar climate and vegetation, supported the overall ability to realistically predict annual and seasonal NEE fluxes (**Figure 79**). The following sections (and Appendix 6) explain the up-scaling process and parameter estimates in more detail.

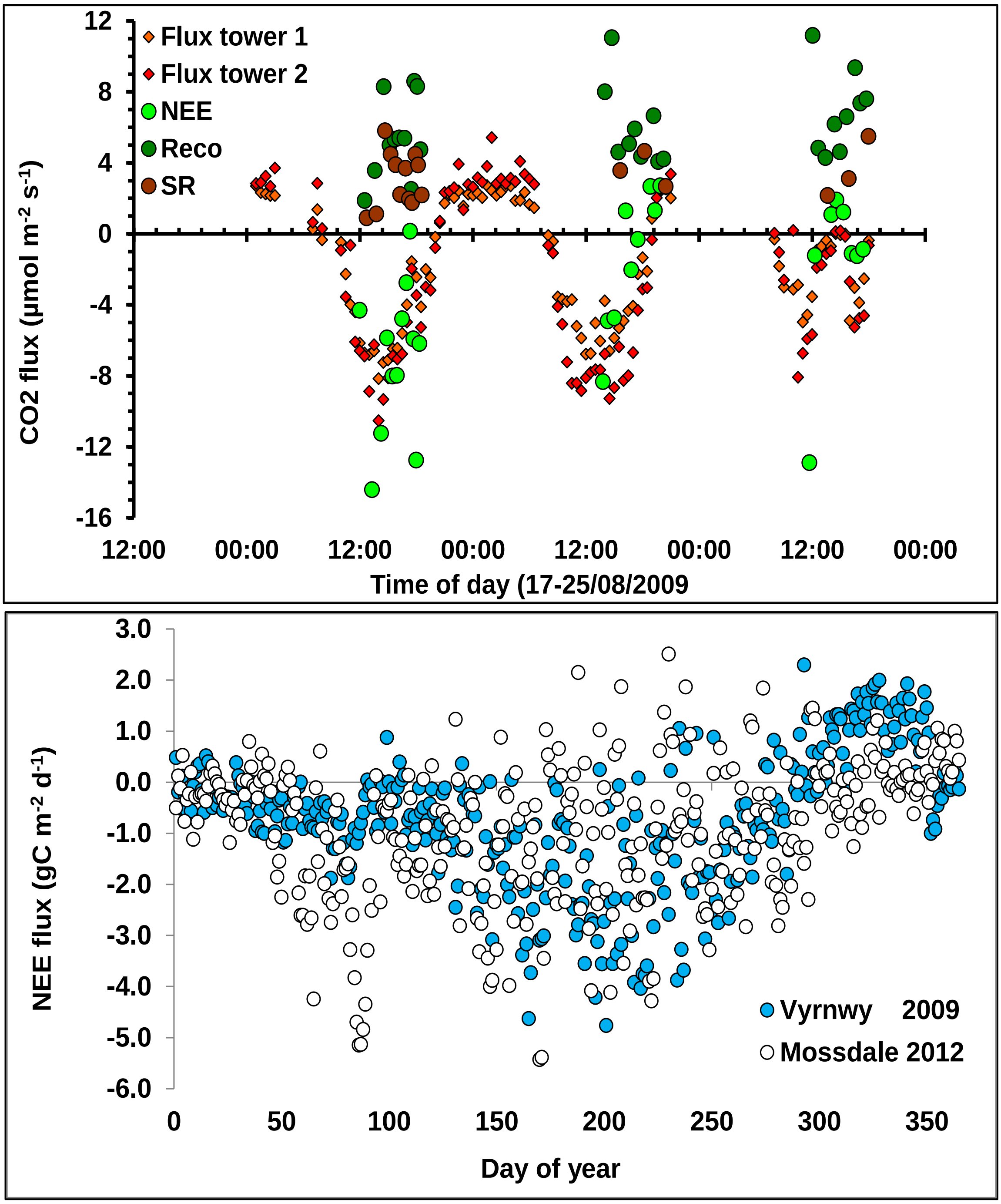


Fig. 79 Comparisons of net ecosystem exchange (NEE). **Top**: fluxes from two continuous Eddy Covariance (EC) systems (Flux tower 1 and Flux tower 2) versus manual light (NEE), dark (R_{eco}) and soil respiration (SR) chamber flux measurements from the EC sites at Vyrnwy (Wales) in August 2009. **Bottom**: fluxes from the Vyrnwy EC system in 2009 and the Mossdale up-scaled NEE fluxes for 2012 based on light response curves obtained in the present study. Note: Mossdale had a similar average climate in 2012 to that at Vyrnwy in 2009 but with a brighter spring, wetter summer and colder winter.

Firstly, measured NEE fluxes for the combined chamber flux measurements for each main management (i.e. n = 4 for uncut (DN), burnt (FI) and mown with brash left (LB)) were regressed against photosynthetically active radiation (PAR) light levels using a non-linear (hyperbolic) curve fit in Excel (**Figure 80**) following Brown (2001). This provided the key model parameters: R_{eco} (respiration in the dark), Km (PAR-to-flux slope) and NEEmax (maximum NEE uptake in the light). PAR levels measured inside the chamber were used, to account for any changes inside the chamber compared to the ambient light conditions measured by the automated weather station (AWS) used for up-scaling fluxes. The same method was applied to all chamber flux data over time and

individual equations for each measurement campaign and their parameters were combined to provide annual equations predicting parameters based on AWS data. Subsequently, the individual PAR-to-flux slopes for the fitted curves were regressed against monthly AWS PAR levels before measurements (**Figure 80**). Only the LB management was included as this was the catchment scale mowing management and there was no significant difference in either NEE light or R_{eco} dark fluxes during the post period (in comparison to brash removal (BR)).

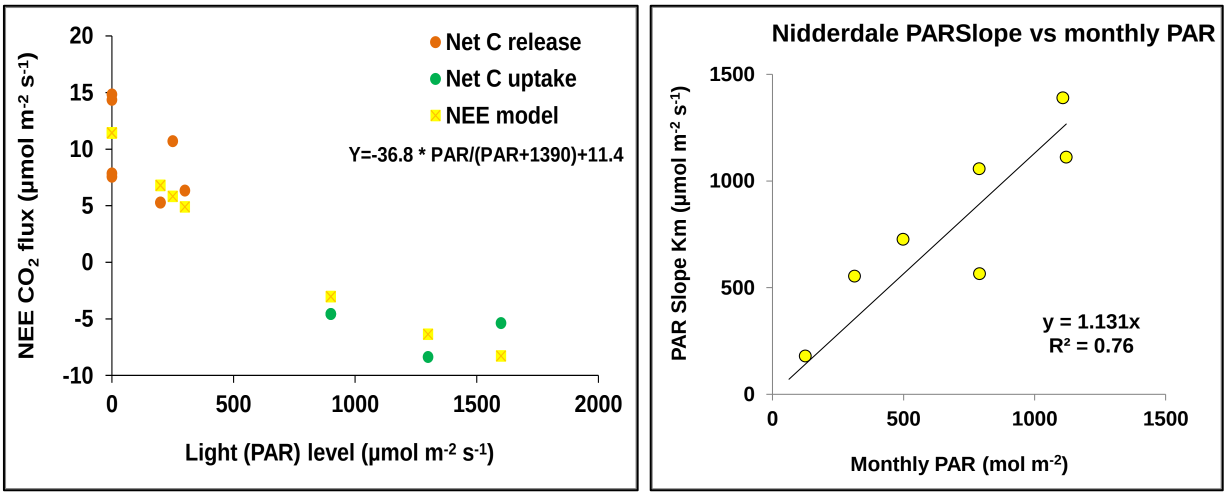


Fig. 80 Examples of (**left**) a chamber net ecosystem exchange (NEE) light response curve fit (crosses) using the combined replicates (n=4) for the Nidderdale uncut (DN) management measured on 18th July 2016 (circles; and (**right**) of a linear regression model fit of the slope (Km value) parameter of calculated photosynthetically active radiation (PAR-to-flux slope) against monthly PAR, with regression equations and r^2 values shown (**right**).

Regressions were fitted using data from one year at first (pre-management period and for the first two years post management for burnt and mown plots) and subsequently in pairs of years (e.g. 2013-2014 for uncut or 2014-2015 for managed treatments). This reflected the continuation of mature vegetation, and thus stable fluxes, on uncut plots and the slower regrowth with changes in flux values on burnt and mown plots. The resulting equations for PAR-to-flux slopes (see **Table A6.1** in Appendix 6) revealed a good fit overall (the mean r^2 was 0.58 at Whitendale, 0.60 at Mossdale and 0.61 at Nidderdale).

Secondly, the individual parameters for R_{eco} and NEEmax obtained for each measurement campaign were regressed against the air temperature at the time of measurement as recorded inside the chamber. This approach took time of day into account and allowed chamber temperature to be related to ambient temperatures measured by the automated weather station, which were used for upscaling in time. An exponential fit was chosen for R_{eco}, whilst NEEmax was best represented with a linear fit (see examples in **Figure 81**).

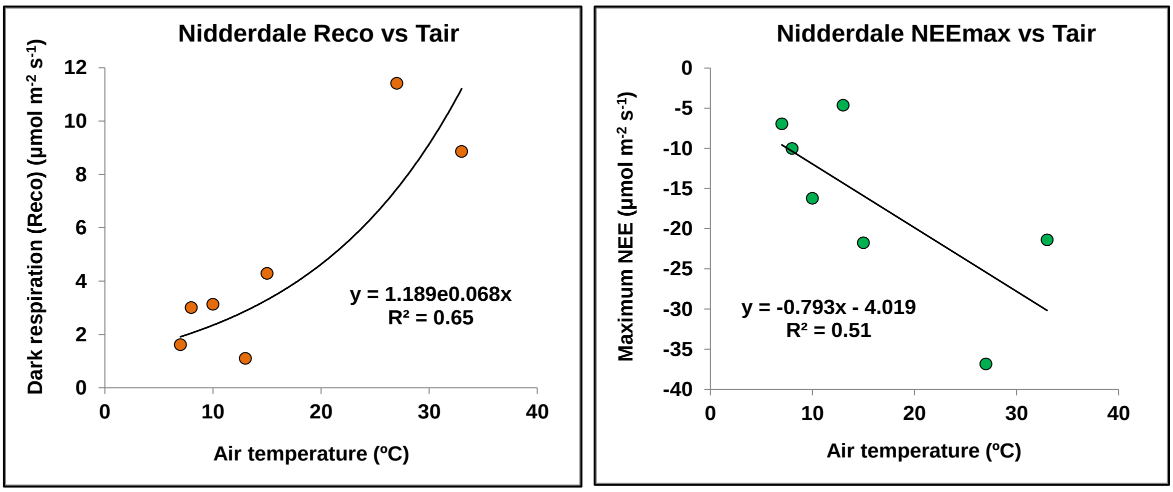


Fig. 81 Regression between air temperature (Tair) in the chamber and (**left**) the fitted dark ecosystem respiration (R_{eco}) values (exponential) and (**right**) the maximum net ecosystem exchange (NEEmax) values (linear) from the light response curve models (n = 7) for all flux measurements at Nidderdale during 2015-2016. Regression equations and r^2 values are shown.

As for the k_m values for the PAR slope, regressions were fitted for one year or in pairs of years reflecting the continuation of mature vegetation on uncut plots and step changes in regrowth on burnt and mown plots. The resulting equations for R_{eco} and NEEmax parameters (see **Table A6.2** in Appendix 6) revealed a good fit overall which was better for R_{eco} than for NEEmax (the mean r^2 for R_{eco} and NEEmax was 0.83 and 0.63 at Whitendale, 0.79 and 0.60 at Mossdale and 0.82 and 0.47 at Nidderdale). Importantly, as for PAR slopes (see **Table A6.1** in Appendix 6), the equation parameters for R_{eco} and NEEmax were fairly stable between years for the uncut management, but showed considerable changes following burning and mowing, with subsequent slow recovery over time to values similar to the uncut management by the final period (2015-2016).

Up-scaled results

Combining the light response curves, PAR slopes and the parameter estimates for R_{eco} and NEEmax enabled prediction of the light compensation point (LCP) over time for each site and measurement campaign (**Figure 82**).

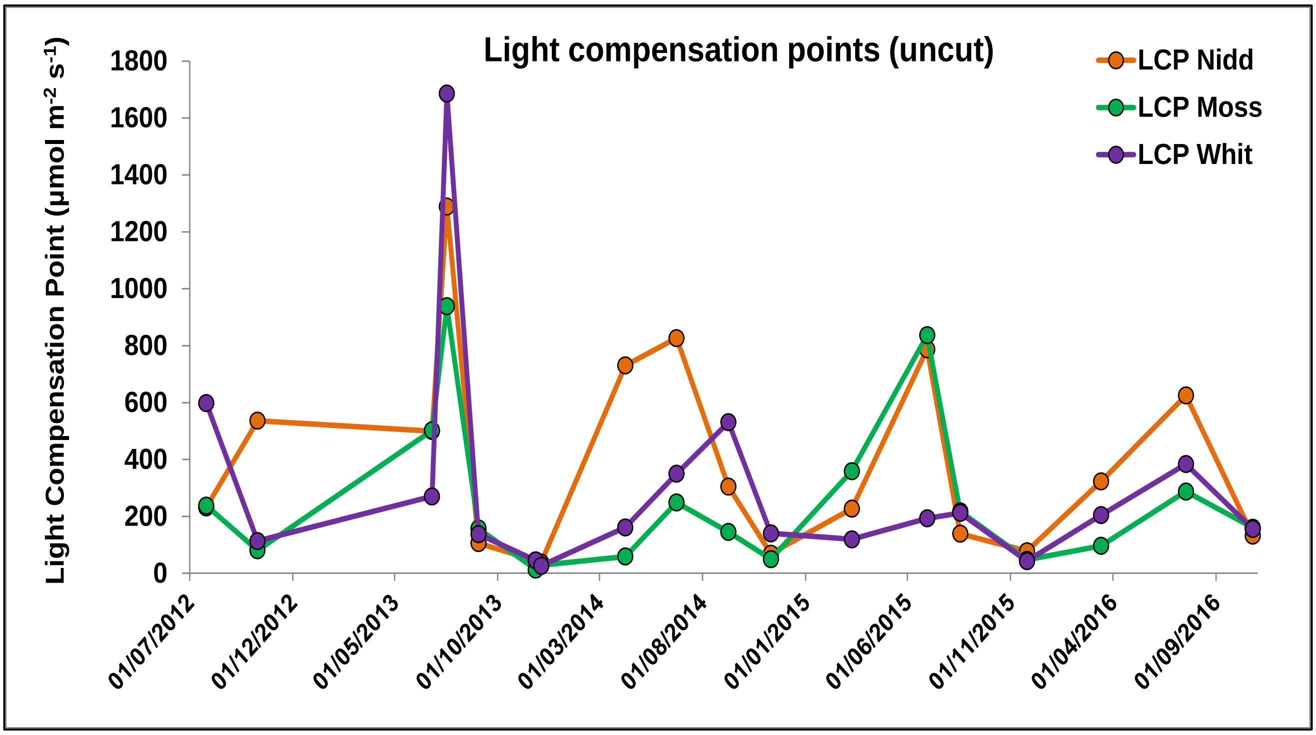


Fig. 82 Predicted light compensation points (LCP; i.e. the light (PAR) level at which a net ecosystem exchange (NEE) flux is zero) over the entire monitoring period (2012-2016) for the uncut (DN) plots at Nidderdale (Nidd), Mossdale (Moss) and Whitendale (Whit).

Notably, this LCP assessment could only be done for the uncut management. Mown and burnt plots did not provide stable estimates, because there were too many fluxes around zero. The light compensation point is an ecologically important parameter as it reveals how efficient vegetation is at taking up carbon, based on identifying the point at which the flux changes from a net carbon source (respiration > photosynthesis) to a net sink (photosynthesis > respiration); a higher LCP indicates that a higher amount of PAR is required to turn the vegetation from a carbon source to a sink. Overall, LCP values were highest during mid-summer after a steady increase during spring and were lowest during winter after a rapid drop in autumn (**Figure 82**). Interestingly, the mean LCP (±SE) was highest at Nidderdale (388±82 µmol m^{-2} s^{-1}), lower at Whitendale (298±89 µmol m^{-2} s^{-1}) and lowest at Mossdale (248±62 µmol m^{-2} s^{-1}). This is consistent with the greater *Calluna* height and *Sphagnum* cover, and greater NEE fluxes (i.e. C uptake) in full light at Mossdale (**Figure 77**).

Assuming an average daily PAR level of 500 µmol m^{-2} s^{-1} with temperatures varying over time, the predicted NEE carbon uptake (NEE500) decreased from Mossdale (NEE500 ±SE; -3.1±0.7 µmol CO_2 m^{-2} s^{-1}) to Whitendale (-1.9±0.5 µmol CO_2 m^{-2} s^{-1}) to Nidderdale (-1.7±0.7 µmol CO_2 m^{-2} s^{-1}). Comparing the estimated LCP to the NEE500 at each site revealed an inverse relationship (**Figure 83**), with Mossdale having the lowest LCP and highest NEE500 carbon uptake, meaning plants there required less light to fix carbon on a net basis and under comparable light levels. Mossdale could therefore be expected to show the highest carbon uptake whereas Nidderdale should likely show the lowest. An explanation might be in the canopy closure or the N-concentrations in the leaves (as greater N content will generally support more photosynthesis). However, biomass seemed to be the main factor as Mossdale had the highest *Calluna* biomass (see **Figure 28** in Section 4.2.5.2) but the lowest N content (see **Table A12.1** in Appendix 12).

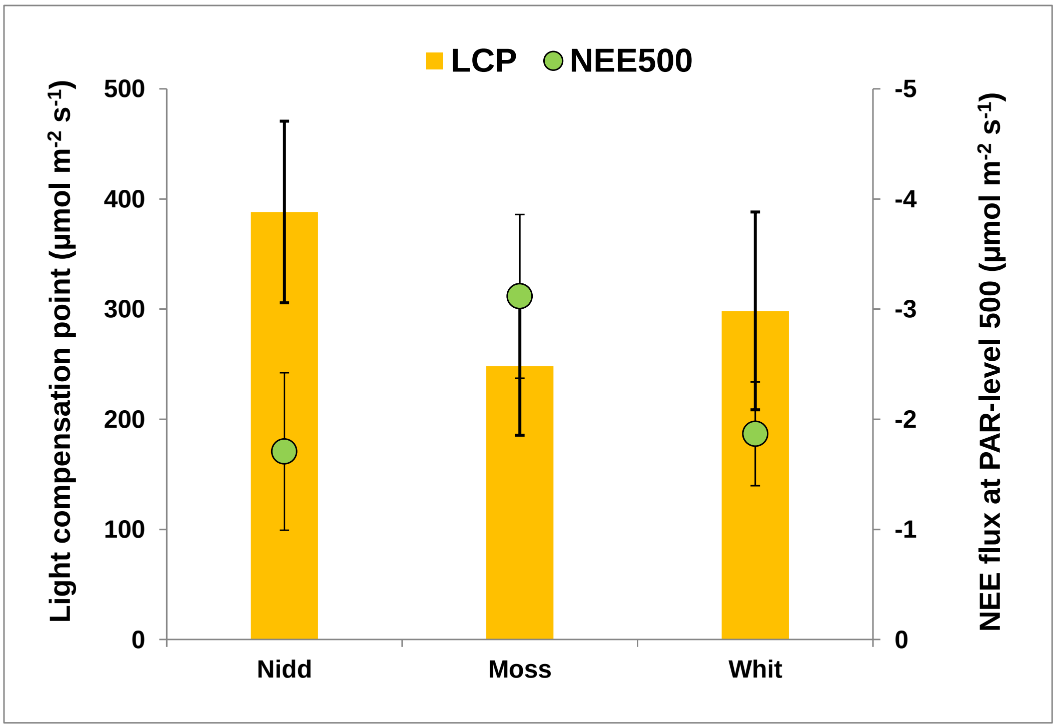


Fig. 83 Average (±SE) light compensation points (i.e. the light (PAR) level at which a net ecosystem exchange (NEE) flux is zero; yellow bars) and the corresponding average predicted NEE fluxes when PAR is 500 µmol m^{-2} s^{-1} (green circles) for the uncut (DN) plots at Nidderdale (Nidd), Mossdale (Moss) and Whitendale (Whit) over the entire monitoring period (2012-2016).

Finally, the overall model NEE C flux predictions, which incorporated hourly AWS temperature data (to predict R_{eco} and NEEmax) and monthly PAR sums (to predict the monthly PAR slope) were compared to the actual flux measurements under full light (i.e. no shaded fluxes were used) during the same time of day (**Figure 84**).

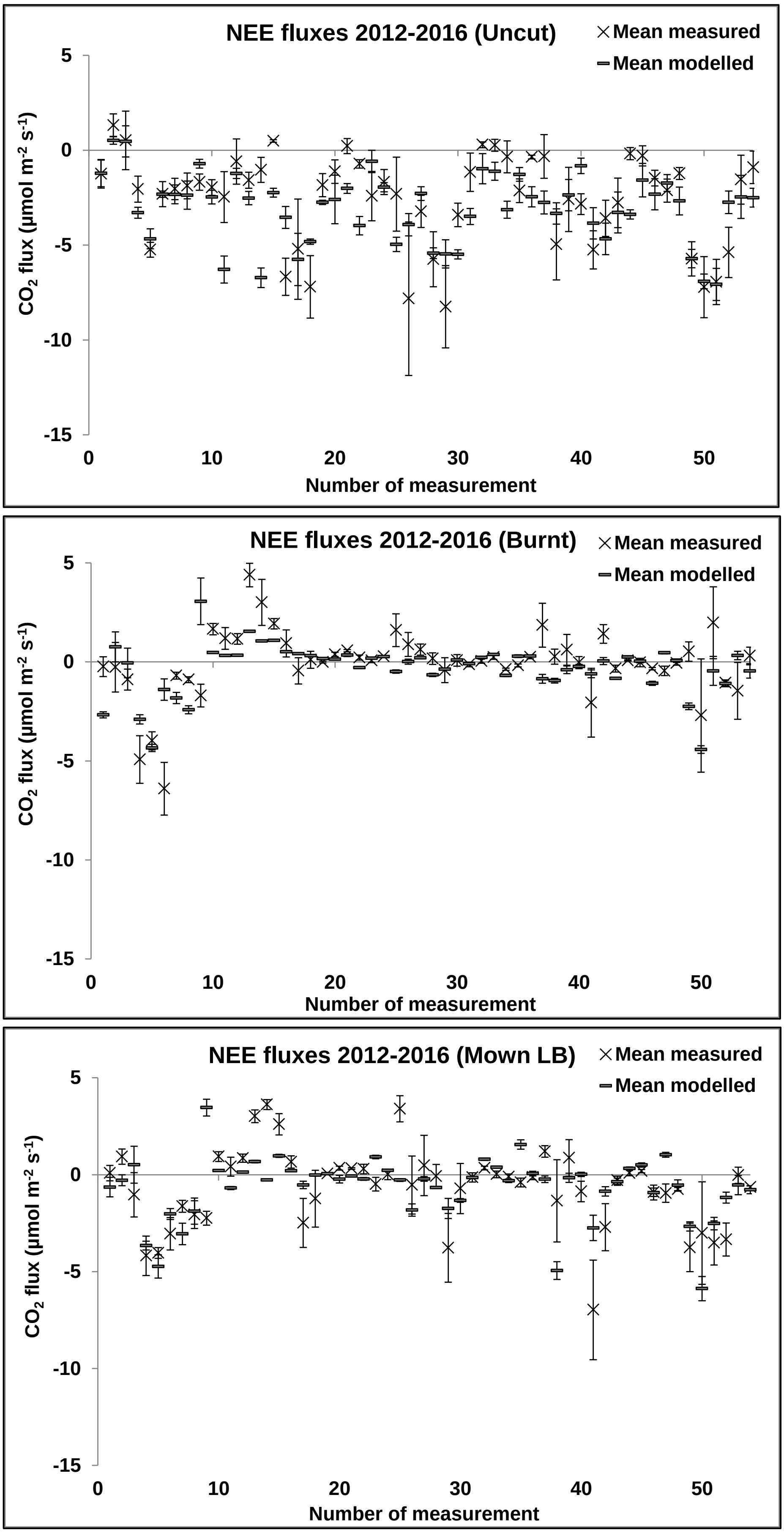


Fig. 84 Comparison of modelled net ecosystem exchange (NEE) CO_2 fluxes for uncut (**top**), burnt (**mid**) and mown with brash left (LB) (**bottom**) based on light response curves and weather station data (using equations shown in **Tables A6.1** and **A6.2** in Appendix 6) with actual chamber NEE flux measurements made during corresponding time periods across all measurement campaigns. The mean values ± 95% CI are shown for the measured values. The onset of management was at measurement 9.

Importantly, although modelled fluxes tended to be closer to zero than the measured fluxes, a paired Student's t-test supported that there was no significant difference between the measured and modelled NEE fluxes (t_{134} = 1.64, p = 0.1043). This confirmed that the model was adequate to be used to upscale the NEE fluxes, for subsequent use in calculation of monthly and annual carbon flux budgets.

A comparison of the monthly NEE carbon flux budgets in the uncut plots showed considerable seasonal changes, with the highest uptake in June and highest losses in December, as well as marked differences between sites (**Figure 85**). **Figure 85** also shows large differences between sites in the annual carbon balance; over the five year period, the highest mean annual carbon uptake on uncut plots was observed for Mossdale (-197 g C m^{-2} yr^{-1}), with Whitendale taking up half the amount (-93 g C m^{-2} yr^{-1}), whilst there were net losses (+19 g C m^{-2} yr^{-1}) at Nidderdale. These data (apart from the drier Nidderdale site) compare well to other carbon budgets for wet blanket bogs obtained with EC flux towers, by Levy & Gray (2015) (-114 g C m^{-2} yr^{-1}) in Scotland, Lloyd (2010) (-173 g C m^{-2} yr^{-1}) at Moor House and Heinemeyer et al. (unpublished) near Lake Vyrnwy in Wales (-264 g C m^{-2} yr^{-1}).

Inter-annual variability was also considerable, with 2012 showing low carbon uptake but 2013 showing high carbon uptake across all sites; Mossdale had the largest difference between 2012 to 2013 of -320 g C m^{-2} yr^{-1}. This is consistent with the wet and dark conditions in 2012 and the bright and warm conditions in 2013 (see **Table 1**). Similar ranges of inter-annual variation in carbon fluxes for blanket bogs have been reported previously for measured (Lloyd, 2010; 70 g C m^{-2} yr^{-1}) and modelled (Worrall et al. 2009; 300 g C m^{-2} yr^{-1}) fluxes for Moor House. However, considerable carbon losses were observed in 2014 after frost damage at Mossdale and Whitendale but not at Nidderdale, where heather plants were protected by snow cover. Interestingly, carbon uptake subsequently recovered in 2015 at Whitendale but not at Mossdale, possibly relating to considerable heather beetle damage in 2015 - visual shoot damage was observed and heather beetles were found in traps used for cranefly surveys. The climate in 2016 was close to the average, with no unusual abiotic or biotic stress, and best reflected the 5-year average carbon flux balance at each site. These data represent the first chamber-based carbon flux balance for actively managed grouse moors on blanket bog in the UK, highlighting variability between sites and years and indicate a longer time frame is required to capture a robust long-term C balance.

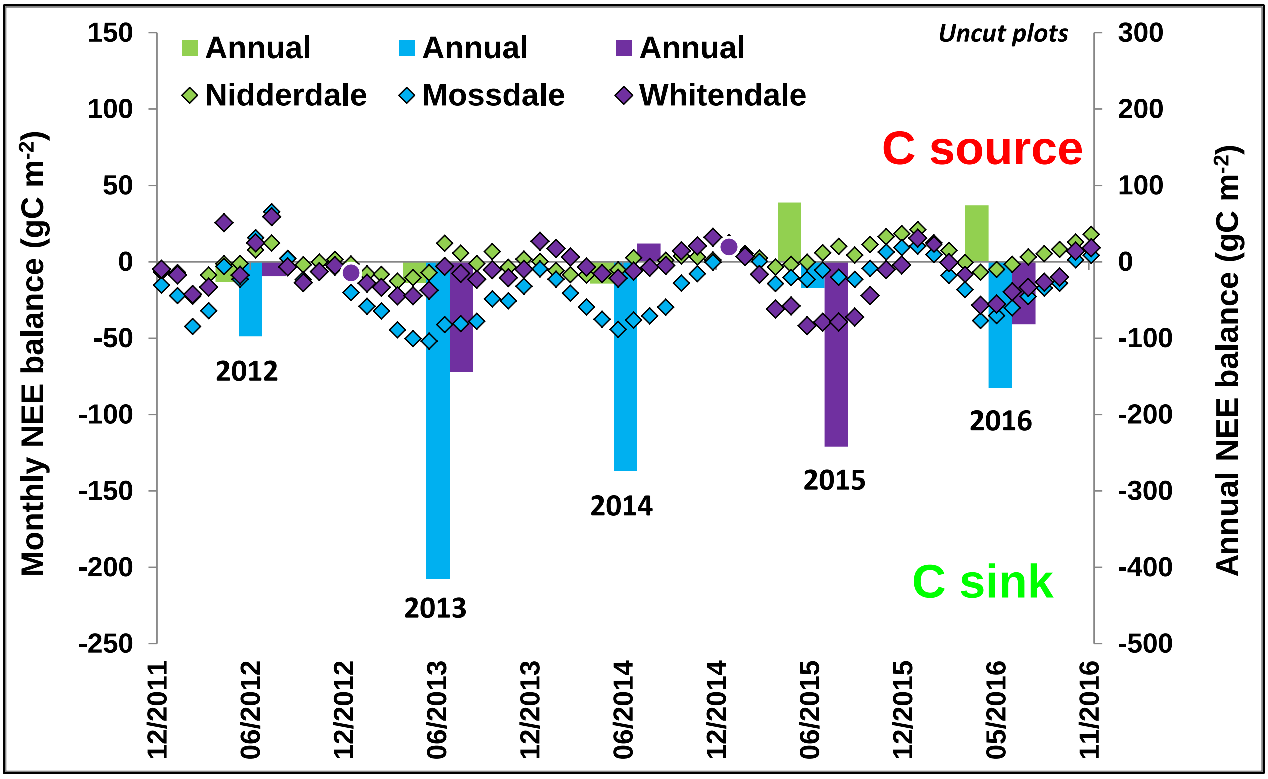


Fig. 85 Monthly (diamonds) and annual (bars) carbon balance totals obtained from modelled hourly net ecosystem exchange (NEE) fluxes for the uncut (DN) management based on light response curves together with hourly weather station data, for the three sites (Nidderdale, Mossdale, Whitendale). Negative numbers indicate net carbon uptake (i.e. a C-sink).

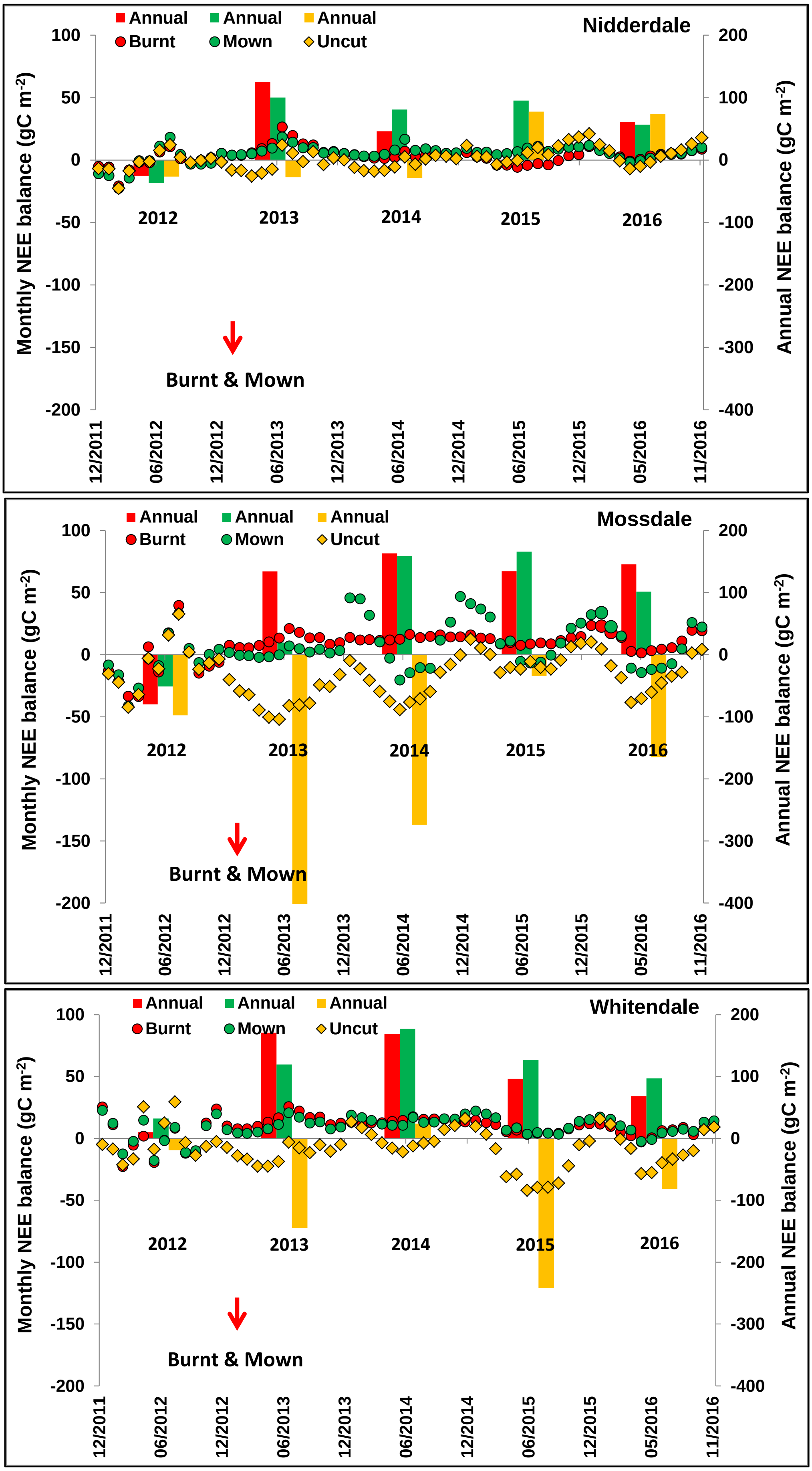


Fig. 86 Monthly (symbols) and annual (bars) C balance totals obtained from modelled hourly net ecosystem exchange (NEE) fluxes based on light response curves together with hourly climate data. Data are shown for the burnt (red), mown with left brash (green) and uncut (orange) managements for all three sites: Nidderdale (**top**), Mossdale (**middle**), Whitendale (**bottom**). Positive numbers represent a C source (i.e. net C loss) and the onset of management is indicated by an arrow.

Management initially showed a similar response at all three sites, with NEE carbon fluxes for burnt and mown plots changing to a net source (**Figure 86** above). Statistics were not possible on those fluxes as for the light response curve modelling the four flux replicates were pooled for each management. However, statistics are provided for the actual measured fluxes in the **Figures 76-78** (above). The difference in the carbon balance of managed plots compared to the uncut plots was initially very large, ranging in 2013 from 128 g C m^{-2} yr^{-1} less carbon uptake at Nidderdale mown plots to 550 g C m^{-2} yr^{-1} less carbon uptake at Mossdale burnt plots, immediately after management. At all three sites, burnt and mown plots switched from being a net C sink to a net C source in 2013, with burnt plots showing a greater loss. However, over time as vegetation regrew, this difference compared to uncut carbon uptake lessened by 2016, ranging from considerably lower mean carbon uptake on combined burnt and mown plots at Mossdale (289 g C m^{-2} yr^{-1}) and Whitendale (165 g C m^{-2} yr^{-1}) to very similar carbon uptake at Nidderdale for the combined burnt and mown plots. Indeed, burnt (in 2015 and 2016) and mown plots (in 2016) at Nidderdale took up more carbon than the uncut plots (2015 and 2016 burnt plots: 85 and 13 g C m^{-2} yr^{-1} higher C uptake, respectively, and 2016 mown plots: 17 g C m^{-2} yr^{-1} more C uptake). Over the four years post-management, the site differences in this lower carbon uptake compared to that of uncut plots was lowest for Nidderdale (mean of 46 g C m^{-2} yr^{-1}) and largest for Mossdale (mean of 350 g C m^{-2} yr^{-1}). The average annual carbon uptake of burnt and mown plots was very similar (109 versus 108 g C m^{-2} yr^{-1}, respectively) and there were few differences at individual sites.

It is not possible to compare the mown NEE C flux budgets with other studies as, to our knowledge, there are none on blanket bogs or other similar peatlands comparing C fluxes on (uncut to) burning and mowing alternatives. Notably, the overall trajectory of the annual carbon source to sink balance based on NEE fluxes from the burnt and mown areas (**Figure 86**) so far does not show a clear indication of a long-term trend. For example, C losses from mown areas could still become substantially greater than from the burnt areas. This could be because, although *Calluna* regeneration is initially slower after burning than after mowing (see Section 4.2.6.3), new growth is more vigorous and can overtake *Calluna* regrowth of mown areas. Moreover, brash decomposition could increase C flux losses on mown plots over time. Whereas all sites initially showed higher net carbon loss on burnt plots, Mossdale subsequently showed similar losses on mown and burnt plots, but both Nidderdale and Whitendale had on average less C losses on burnt than on mown plots from 2014 onwards. At all three sites, this difference in net carbon uptake probably related to differences in re-growth and cover of *Calluna* and other vegetation (see **Figure 37**), as burning has been shown to significantly affect other peatland species (Harris et al., 2011), particularly bryophytes (Burch, 2008). Clearly longer timescales are needed to capture the long-term trends and actual differences between managements. So far, vegetation regrowth has not reached a steady state, which is needed to gain confidence in the long-term carbon budgets. As at these sites heather reaches near canopy closure at around 10 years (i.e. height of ~30 cm) five more years of monitoring should better capture a long-term trend.

The combination of measured NEE (**Figure 84**) and SR component fluxes (**Figure 72**) in 2015 and 2016 allowed the 'break-down' of the total flux for uncut plots (specifically so as to compare it to literature values), providing information on respiration components and key carbon cycle parameters (see Heinemeyer et al., 2012 for detailed methods), namely gross primary productivity (GPP), net primary productivity (NPP) and carbon use efficiency (CUE) and their ratios (see **Figure A6.4** in Appendix 6). Notably, the ratio of root (Rr) to total soil respiration (Rs) based on the vegetated NEEnew plots (Ra) was on average 0.56, which was considerably higher than the root contribution of 0.30 based on the vegetation-free areas (**Figure 73**), reflecting higher root biomass in undisturbed areas. Carbon flux components were highest on uncut, lower on mown and lowest on burnt plots with the ratios being very similar to one another (see **Figure A6.4** in Appendix 6). Whilst the R_{eco}/GPP ratio was higher on burnt and mown plots, CUE was very similar for all managements and only GPP and NPP were higher at Mossdale. A similar analysis for the other management types can only be done once the vegetation has re-grown sufficiently (i.e. for large sections, the fluxes were still too small to be meaningfully different from zero).

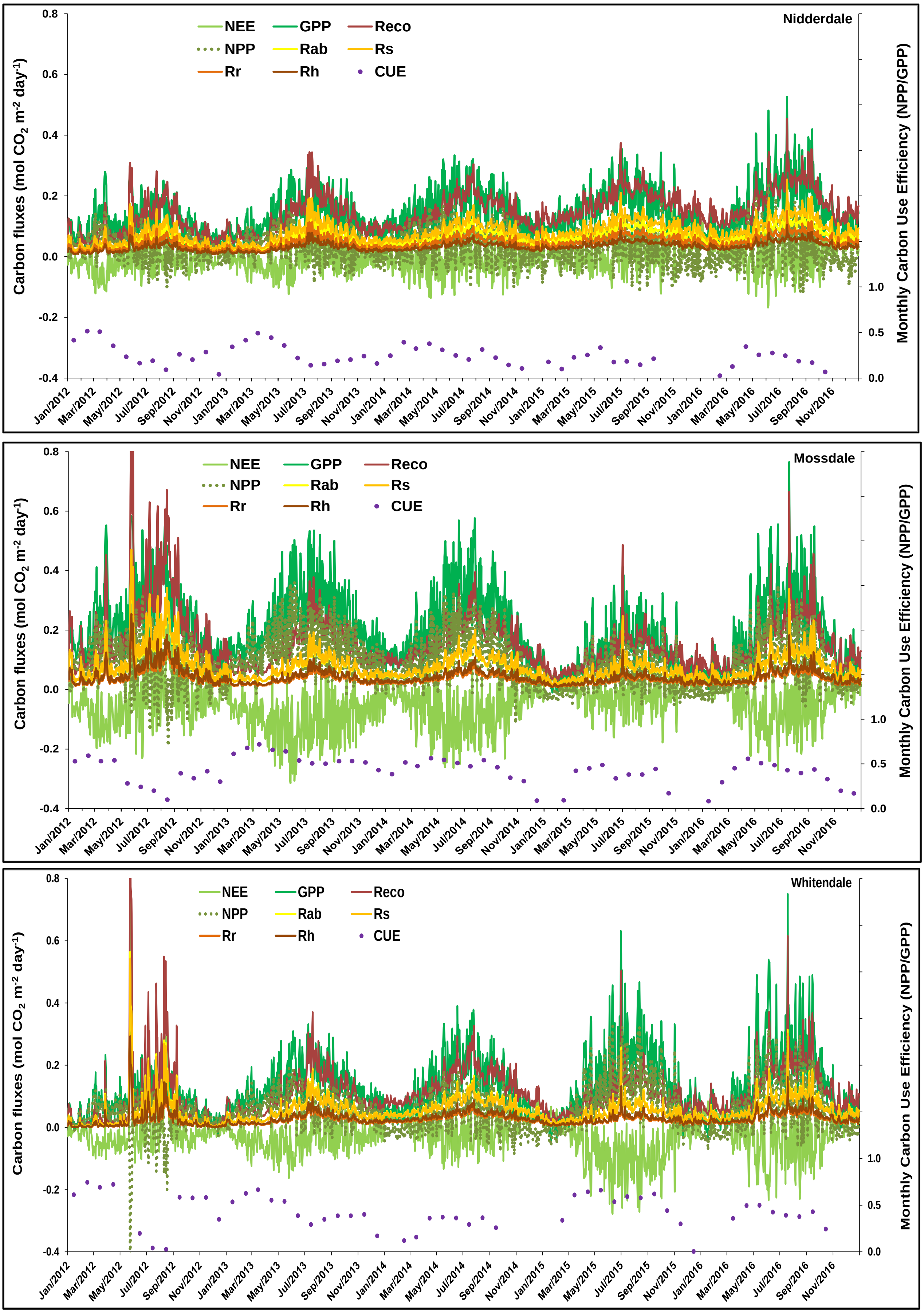


Fig. 87 Daily sums of the major carbon fluxes (NEE, GPP, R_{eco}) based on light response curves using climate parameter regressions (see Appendix 6), predicted net primary productivity (NPP) and respiration flux components (above ground: Rab, total soil: Rs, root: Rr and decomposition: Rh) obtained from modelled hourly net ecosystem exchange (NEE) fluxes. For the approach for site specific average ratios see **Figure A6.4**). Data show the uncut (DN) plot averages together with monthly carbon use efficiencies (CUE) (average NPP/GPP) for Nidderdale (**top**), Mossdale (**middle**) and Whitendale (**bottom**).

Combining the ratios of the uncut carbon flux components and the key carbon cycle parameters (see **Figure A6.4** in Appendix 6) with the modelled NEE fluxes (**Figure 85**) enabled estimation of daily flux components for the uncut plots. The predicted daily flux components (**Figure 87** above) indicated clear seasonal patterns, with both the greatest C uptake (GPP, NPP) and greatest C loss (R_{eco}) during the summer, resulting in the highest CUE in spring and autumn. Overall, flux components and ratios varied seasonally and between sites, with summers showing the highest variation and CUE declining in the order of Mossdale to Whitendale to Nidderdale.

Calculated annual totals for uncut plots (**Table 15**) showed large inter-annual variability and marked site differences, with greatest net carbon uptake (NPP) for Nidderdale and Mossdale in 2013 and 2014, although GPP was highest in 2012 at Mossdale when R_{eco} was also highest. Notably, 2014, and particularly 2013, had high annual PAR and low rainfall (**Table 1**). However, NPP at Whitendale was relatively low in 2014, possibly due to frost damage (see Section 3.6). The overall mean NPP and CUE were highest at Mossdale (350 g C m^{-2} yr^{-1} and 0.34, respectively). Carbon losses (R_{eco} and its components) were much higher at Mossdale and Nidderdale than at Whitendale.

Table 15 Yearly and mean total annual carbon fluxes (g C m^{-2}), average carbon use efficiency (CUE) and yearly mean totals for individual respiration components for each of the three sites for uncut plots during 2012-2016 Data are based on values plotted in Fig. 84 (based on flux parameters and carbon ratios shown and explained in Fig. A6.4 in Appendix 6).

Site	Year	NEE gC m^{-2}	Reco gC m^{-2}	GPP gC m^{-2}	NPP gC m^{-2}	CUE (NPP/GPP)	Rab gC m^{-2}	Rs gC m^{-2}	Rr gC m^{-2}	Rh gC m^{-2}
Nidderdale	**2012**	-26.5	449.8	476.4	129.8	0.27	197.9	251.9	148.6	103.3
	2013	-27.5	541.5	569.0	151.8	0.28	238.3	303.2	178.9	124.3
	2014	-28.5	639.9	668.4	175.4	0.24	281.6	358.4	211.4	146.9
	2015	77.7	770.9	693.2	99.3	0.11	339.2	431.7	254.7	177.0
	2016	73.9	794.3	720.4	108.5	0.10	349.5	444.8	262.4	182.4
	Mean	**13.8**	**639.3**	**625.5**	**133.0**	**0.20**	**281.3**	**358.0**	**211.2**	**146.8**
Mossdale	**2012**	-97.8	969.8	1067.6	311.1	0.32	484.9	484.9	271.6	213.4
	2013	-415.6	604.0	1019.6	548.4	0.54	302.0	302.0	169.1	132.9
	2014	-274.1	705.2	979.3	429.2	0.39	352.6	352.6	197.5	155.1
	2015	-34.1	485.8	519.8	140.9	0.13	242.9	242.9	136.0	106.9
	2016	-165.3	698.7	864.0	319.0	0.31	349.4	349.4	195.6	153.7
	Mean	**-197.3**	**692.7**	**890.1**	**349.7**	**0.34**	**346.4**	**346.4**	**194.0**	**152.4**
Whitendale	**2012**	-19.1	371.4	390.4	113.3	0.41	170.8	200.5	106.3	94.3
	2013	-144.6	461.6	606.2	261.7	0.43	212.3	249.3	132.1	117.2
	2014	23.9	594.0	570.1	126.8	0.10	273.2	320.8	170.0	150.8
	2015	-242.2	399.1	641.3	343.5	0.08	183.6	215.5	114.2	101.3
	2016	-82.0	590.0	672.0	231.8	0.21	271.4	318.6	168.8	149.7
	Mean	**-92.8**	**483.2**	**576.0**	**215.4**	**0.25**	**222.3**	**260.9**	**138.3**	**122.6**

Previous work in an oak forest using a similar approach (but utilising EC flux tower data together with soil component fluxes) provided an annual breakdown of the carbon flux components and main parameters (Heinemeyer et al., 2012), but the present study provides the first such data for a blanket bog. Only one other blanket bog study exists that included soil respiration data alongside a flux component comparison based on eddy covariance fluxes, which was in a PhD thesis with data from Moor House (Lloyd, 2010). The study by Lloyd (2010) estimated annual carbon fluxes for NEE (-173 g C m^{-2} yr^{-1}), R_{eco} (718 g C m^{-2} yr^{-1}), GPP (891 g C m^{-2} yr^{-1})*, Rab (457 g C m^{-2} yr^{-1}) and Rs (262 g C m^{-2} yr^{-1}); applying the same root respiration fraction of 56% (as used in **Table 15**) allowed us to also estimate annual values for Rr (144 g C m^{-2} yr^{-1}), Rh (118 g C m^{-2} yr^{-1}) and consequently NPP (298 g C m^{-2} yr^{-1}) and CUE (0.33) for the Moor House site. All values for uncut plots are remarkably similar to the overall average obtained across the three sites, particularly to the Mossdale averages, in this study (**Table 15**). The slightly higher R_{eco} and GPP at Moor House could reflect the older heather community (i.e. project sites with ~20 year old heather versus Moor House NNR with no burn rotation since 1950 and thus likely >40 year old *Calluna* with uneven stem age classes reflecting possibly natural layering/re-growth through a moss layer). However, Levy & Gray (2015) measured a GPP of only 575 g C m^{-2} yr^{-1}, R_{eco} of 461 g C m^{-2} yr^{-1} and NEE of 114 g C m^{-2} yr^{-1} at Moor House, values that are more similar to the overall average found in this study. **Note that in Lloyd (2010) the value for GPP is wrongly stated as 457 g C m^{-2} yr^{-1} (in Figure 7-1).*

4.2.15 *Greenhouse gas emissions*

Method summary

Greenhouse gas (GHG) fluxes were measured ~4 times a year on a combination of plots (i.e. with or without vegetation; see Appendix 8 for more detailed methodological information and a summary of data analysis methods and **Table A6.1** in Appendix 6 for flux measurement dates). The moss surface was removed from a 20 cm diameter circle near the water table depth logger to create a bare monitoring area (**Figure 88**) which contained no vascular plants. This created a small indentation (which did not alter the peat surface level) which enabled exact re-location, as well as removing the surface autotrophic C fluxes to obtain comparable decomposition emissions. However, brash and litter was left in place on all measurement areas to assess management impacts.

Fig. 88 Greenhouse gas (GHG) flux measurements using either a manual syringe method (2012-2014) for later laboratory gas chromatography analysis (far left), including manual temperature monitoring for gas concentration correction (2nd from left), or an instantaneous in situ method (2014-2016) using a Los Gatos GHG ultraportable field analyser (two pictures on the right) allowing for real time monitoring of fluxes on a wireless handheld device (far right). Neither method used any soil collars inserted into the peat but both used a ring of removable *Sphagnum* moss to seal the collar or chamber base.

During field visits, any plants growing within the plot areas were removed. Methane (CH_4) and nitrous oxide (N_2O) fluxes, as well as NEE and SR, were always measured on the same day at a single site in order to reduce climate variations within sites in a single measurement set. Over the five year period, the methods for measuring GHG emissions changed. Measurements in 2012 and 2013 were made using cover-boxes (20 cm diameter, 10 cm tall collar and 15 cm tall chamber), which were based on a modification of the static chamber method described in Livingston & Hutchinson (1995), from which 20 ml gas samples were withdrawn at set time points (generally four samples over 50 minutes) and stored in evacuated 12 ml glass vials (Labco, UK). These samples were then analysed for CH_4 and N_2O concentrations on a laboratory-based gas chromatograph (PerkinElmer ARNEL AutoSystem XL gas chromatograph; PerkinElmer Instruments, Shelton, USA). The chambers were insulated with reflective wadding to minimise any internal temperature change. Internal chamber temperatures were monitored and, even on hot days, changed by less than 3°C during the whole closure period. The open end of a sealed 2 m length of vacuum tubing (Tygon, Akron, USA) was pushed through a rubber bung (SubaSeal, Sigma-Aldrich, USA) which was secured in a small hole drilled through the top of each chamber and thus allowed gas sampling with a hypodermic needle (**Figure 88**). Collars were placed on the sample area and sealed with wet *Sphagnum* moss, which was packed around the base to create an airtight seal. Chambers, with an open vent, were then placed on top and sealed by means of a 10 cm wide rubber band rolled over the join and a rubber bung at the top. N_2O fluxes were only measured during 2012 and 2013; since the fluxes were very low and close to the detection limit, no further measurements of N_2O were made.

CH_4 measurements from June 2014 onwards were measured in real time using an Ultraportable Greenhouse Gas Analyser (UGGA; Los Gatos Research, USA). The UGGA was connected to an insulated chamber with two sections of tubing (Bev-a-line IV, Thermoplastic Processes, Inc., USA), creating a closed system which was fitted with a LiCor vent (**Figure 88**). The chamber was placed gently onto the plant-free measurement area and sealed with wet *Sphagnum*. From June 2015 onwards, methane emissions were also measured over the burnt or cut NEE plots with re-growing vegetation, with vegetation cover also recorded, which required a 30 cm diameter chamber. A tablet (Google Nexus, USA) was used to view the flux in real time. On plots where fluxes spiked and then dropped (indicating a bubble), the chamber was removed, vented and replaced. However, some consistently very high

fluxes were recorded, particularly at Mossdale (the wettest site) and most likely this reflected ebullition events (either natural or due to chamber placement). Fluxes were generally measured for 2 - 5 minutes. As the method for CH_4 flux measurement changed half way during the project phase, two comparisons were done in 2014 (at Nidderdale in April and at Mossdale in June). For this, both the cover box gas sample GC-method and the in-situ flux UGGA-method were used over the same plots. The comparison (see **Figure A8.1** in Appendix 8) showed an overall good agreement between mean fluxes, particularly for the lower fluxes measured in April, although there were greater differences between the two methods for June samples.

To assess the impact of vegetation, and particularly sedge, cover on measured CH_4 fluxes, fluxes for the bare soil respiration plots were compared to fluxes measured on the vegetated (re-growing) NEEnew plots (see **Figure A8.2** in Appendix 8). Fluxes were higher from vegetated areas, even when the very high emission plots at Mossdale were excluded (see ebullition comment above). Therefore, statistical analysis for all vegetated plots included the percentage cover of sedge within the NEE chamber area, which was recorded to the nearest 5% each year based on close-up photographs, with a Perspex chamber collar marking the flux area; see **Figure 74**. Further details of the methods, including those used for up-scaling, are provided in Appendix 8.

Results

The CH_4 fluxes were very variable, ranging from -137.6 to 7145 nmol CH_4 m^{-2} s^{-1}, with a mean of 77.2 nmol CH_4 m^{-2} s^{-1}. However, over 25% of the fluxes were 0 nmol CH_4 m^{-2} s^{-1} meaning that the median flux was only 3.2 nmol CH_4 m^{-2} s^{-1}. There were only 10 fluxes (out of 1368) which had values over 2000 nmol CH_4 m^{-2} s^{-1} and these may have included undetected but likely natural ebullition; these fluxes were included in the dataset used for upscaling and analysis, but emissions were also calculated excluding those data points (see Appendix 8). The CH_4 fluxes were up-scaled for all managements across each site, using time only, as there was no significant overall relationship with any environmental variable. A recent study which compared modelled CH_4 fluxes using environmental factors to CH_4 fluxes simply up-scaled using time (Green & Baird, 2017) revealed only small differences overall, although it did not provide any comparison of modelled versus measured fluxes. However, frequent or extended manual measurements would clearly allow a more robust upscaling approach based on more detailed regression models including robust information on environmental factors explaining flux variability.

There was a considerable difference in up-scaled (annual) fluxes between sites and years (**Table 16**). Overall mean annual CH_4 emissions across all managements and years (including very high emissions in 2015 and 2016) were 11±1 g C m^{-2} yr^{-1} at Nidderdale and 14±2 g C m^{-2} yr^{-1} at Whitendale. These values for Nidderdale and Whitendale were slightly higher compared to published values for the Flow Country (4.3 g C m^{-2} yr^{-1}; Levy & Gray, 2015) and Moor House (3.9 g C m^{-2} yr^{-1}; Worrall et al., 2007, and 6.4 g C m^{-2} yr^{-1}; Worrall et al., 2009). The mean value of 72±9 g C m^{-2} yr^{-1} at Mossdale was considerably higher than any previously reported values for Moor House but close to the upper range of those reported by Cooper et al. (2014) of 54±8 g C m^{-2} yr^{-1} for *Eriophorum* dominated infilled ditches. However, the high Mossdale emissions reflected particularly high fluxes in 2016. Importantly, the latest data from Moor House in 2016 (unpublished; pers. comm. Rob Rose; CEH) covered a similar flux range (2 to 400 g C m^{-2} yr^{-1}) as observed here (**Table 16**), and the total annual emissions (from plots with similar vegetation cover and peat depth as in this study) were between 2 and 10 times those recorded previously, similar for uncut plots to our study (**Table 16**). Notably, 2016 was the second warmest year, had high light (PAR), and was wet overall (**Table 1**); it also followed the record 'wettest winter' in north west England in 2015/16 (**Figure 10**). Moreover, such high fluxes from bogs of up to 60 g C m^{-2} yr^{-1}, particularly in relation to a combination of environmental factors, has recently been reported by Abdalla et al.'s (2016) review. Therefore, these data over two years for three sites could show the likely future trajectory of methane emissions from peatlands under the predicted scenarios of warmer summers following wetter winters. Overall mean annual CH_4 emissions across all managements and years were less when excluding the very high emission fluxes above 2000 nmol m^{-2} s^{-1} in 2015 and 2016 (1 at Whitendale and 9 at Mossdale) and those fluxes are shown in the **Table A8.2** in Appendix 8.

Table 16 Up-scaled mean annual CH_4 fluxes (g C m^{-2} yr^{-1}; mean ± standard error) based on manual chamber fluxes taken 3-4 times per year during the pre- (2012) and post-management (2013-2016) period with the values combined for mown plots with and without *Sphagnum* addition. Whilst in 2012-2014 only fluxes from non-vegetated areas were available, in 2015 and 2016 CH_4 fluxes were measured from the re-vegetating plots from the same areas that the NEE fluxes were measured. Data include 10 fluxes out of 1368 (1 at Whitendale and 9 at Mossdale), which had values over 2000 nmol CH_4 m^{-2} s^{-1}.

Nidderdale Code	Treatment	2012 Pre-management	2013 Post-management	2014	2015	2016	2013-2016 Post-management combined
FI	Control (Burnt)	3.1 ± 2.3	0.2 ± 0.1	1.6 ± 0.6	2.3 ± 0.7	9.4 ± 4.8	3.4 ± 0.8
DN	Do nothing (uncut)	0.1 ± 0.1	1.1 ± 0.8	4.0 ± 3.4	4.4 ± 1.6	73.7 ± 26.6	20.8 ± 4.1
BR	Mown, brash removed	1.7 ± 0.9	1.6 ± 0.8	6.5 ± 3.2	15.0 ± 5.7	4.7 ± 1.4	7.0 ± 1.4
LB	Mown, brash left	1.6 ± 1.3	1.2 ± 1.3	49.2 ± 21.1	11.0 ± 3.9	28.8 ± 15.8	22.5 ± 5.3

Mossdale Code	Treatment	2012 Pre-management	2013 Post-management	2014	2015	2016	2013-2016 Post-management combined
FI	Control (Burnt)	5.4 ± 3.1	3.7 ± 1.2	58.3 ± 48.9	12.6 ± 4.4	196.9 ± 125.2	67.9 ± 22.5
DN	Do nothing (uncut)	15.7 ± 8.1	6.6 ± 3.1	36.7 ± 12.6	164.8 ± 97.6	552.9 ± 290.4	190.2 ± 50.5
BR	Mown, brash removed	13.9 ± 5.3	2.0 ± 2.5	65.5 ± 25.1	134.3 ± 46.6	119.8 ± 78.5	80.4 ± 19.1
LB	Mown, brash left	3.1 ± 1.2	1.6 ± 1.0	15.5 ± 6.9	25.6 ± 9.6	13.6 ± 3.1	14.1 ± 2.6

Whitendale Code	Treatment	2012 Pre-management	2013 Post-management	2014	2015	2016	2013-2016 Post-management combined
FI	Control (Burnt)	0.3 ± 0.2	0.1 ± 0.1	0.9 ± 0.4	25.8 ± 19.1	72.0 ± 68.8	24.7 ± 11.0
DN	Do nothing (uncut)	3.7 ± 2.1	1.2 ± 0.4	2.4 ± 0.8	17.3 ± 4.9	21.3 ± 8.4	10.6 ± 1.8
BR	Mown, brash removed	3.7 ± 3.0	0.6 ± 0.4	18.2 ± 7.2	31.0 ± 9.6	32.7 ± 15.0	20.6 ± 4.0
LB	Mown, brash left	-0.1 ± 0.1	0.9 ± 0.3	14.1 ± 7.5	17.3 ± 4.0	13.7 ± 3.8	11.5 ± 1.9

Overall, there was no significant interaction between management and time period ($F_{5,\ 1239}$ = 1.33, p = 0.25), nor was there any between management, time period and site ($F_{10,\ 1239}$ = 1.34, p = 0.21). When the data were analysed separately for each measurement method, there was no significant interaction between time period and management for the GC-analysed data ($F_{5,\ 542}$ = 1.26, p = 0.28), nor between managements for the non-vegetated UGGA-analysed data (for which there was no pre-management period; $F_{5,\ 265}$ = 0.79, p = 0.56). However, there was a significant difference between managements for UGGA measurements (2015-2016; **Table 16**) from vegetated ground ($F_{5,\ 408}$ = 4.06, p = 0.0013), with CH_4 fluxes from DN (uncut) plots (139 g C m^{-2} yr^{-1}) being significantly higher than those from FI (burnt) plots (53 g C m^{-2} yr^{-1}). This illustrates the importance of both including vegetation in monitored areas and in long term monitoring, since these differences were only found more than two years after the onset of management. However, variability was considerable as shown in the comparison of CH_4 fluxes including the ±*Sphagnum* pellet additions (i.e. which otherwise were combined as they

did not show any measureable pellet establishment or difference in vegetation cover) for each mown management (**Table A8.1 and A8.3** in Appendix 8, considering averages reflecting with or without the very high methane fluxes).

Although our study did not find a significant overall management impact, the significant management effect from the measurements on vegetated areas corroborates the findings of Ward et al. (2007) who measured lower CH_4 fluxes on burnt plots than on unburnt plots. Unfortunately, there are no other studies comparing CH_4 emissions on blanket bogs which have been managed by mowing. However, as CH_4 emissions from the different mowing managements at all sites spanned such a wide range of values, which encompassed all DN and FI values, it appears that mowing did not consistently affect CH_4 fluxes.

Despite the similarity of fluxes between managements for the majority of the study, there were significant differences in (square root transformed) CH_4 fluxes between sites for measurements made by both methods on non-vegetated areas ($F_{2,\ 542}$ = 17.8, p< 0.0001 and $F_{2,\ 265}$ = 6.13, p = 0.0025 for GC and UGGA methods, respectively) and overall ($F_{2,\ 7}$ = 6.64, p = 0.0221), with Mossdale (72 g C m^{-2} yr^{-1}) releasing significantly more CH_4 (about 6 times higher as mean flux) than either Nidderdale (11 g C m^{-2} yr^{-1}) or Whitendale (14 g C m^{-2} yr^{-1}) (**Table 16**). As Mossdale was generally established to be the wettest site (i.e. higher precipitation and WTDs, often with areas of standing water), WTD was deemed to be the most likely factor influencing CH_4 fluxes. This was indeed the case for CH_4 fluxes overall and was supported by the significant positive relationship between CH_4 fluxes and the WTD averaged over the four weeks before measurements, with high WTDs increasing CH_4 emissions (coefficient = 0.052; $F_{1,\ 1249}$ = 8.83, p = 0.0030; **Table A6.4**, Appendix 6). In addition, soil temperature (Tsoil) also had an overall marginally significant (Tsoil: p-value = 0.0352; $Tsoil^2$: p-value = 0.0512) positive impact on methane emissions (see **Table A6.4**). However, the relationships were not robust enough to be included in upscaling. This is consistent with the reported effects of WTD and soil temperature in other studies of methane emissions in wetlands (Hargreaves & Fowler, 1998; MacDonald et al., 1998; Lai et al., 2014). Vegetation has also been shown to have a strong association with CH_4 fluxes (Couwenberg et al., 2011; Gray et al., 2013; Lai et al., 2014). In our study, as both burning and mowing greatly altered the surface vegetation around the measurement areas, it was expected that this would also alter CH_4 fluxes. Although it was not possible to test the influence of vegetation on vegetation-free plots, there was a significant effect on the vegetated plots of the amount of sedge cover on CH_4 fluxes, with higher sedge cover being associated with higher CH_4 emissions, during measurements from 2015 onwards ($F_{1,\ 408}$ =21.1, p< 0.0001; **Table A6.4**). This observation could explain why a management effect was only observed during the latter phase including vegetation but not overall. Moreover, as with NEE and SR fluxes, despite the significant effects of temperature and WTD, the month of measurement also significantly affected the CH_4 fluxes (**Table A6.4**), with greatest fluxes observed in August. However, although the inclusion of these environmental effects could improve the modelled fluxes, additional data are needed including on vegetated plots as so far relationships were not robust enough.

The N_2O fluxes were consistently very low (mean of 3.9 µg m^{-2} h^{-1}) and often near the detection limit, which is in agreement with a range of -4.25 to +9.91 µg m^{-2} h^{-1} reported for a similar study on a Scottish bog (Dinsmore et al., 2009). Fluxes of N_2O did not differ between any sites or managements and there was no temperature or water table effect (see Appendix 8 for data and statistical information). Overall, N_2O fluxes were therefore estimated to average 0.035 g N_2O m^{-2} yr^{-1}.

Notably, these combined GHG emissions (i.e. CO_2 from SR and CH_4 and N_2O net emissions) seem to be the first annual estimates for a heather dominated UK blanket bog under grouse moor management. An important finding was the interannual variability and also differences in and impacts of site wetness. Clearly, as shown in the analysis by Smyth et al. (2015) and highlighting that predominantly short-term studies (i.e. < 5 years) were used, longer-term data are needed to define emissions in relation to the IUCN UK's Peatland Code, to include grouse moor blanket bog areas and to determine whether there is any consistent long-term variability between sites. As Smyth et al. (2015) state "there is a need to encourage and support long term monitoring. The lack of funding for

biological monitoring is the primary barrier to monitoring peatland biodiversity and impacts of restoration". Moreover, so far there is no clear indication as to management impacts on GHG emissions overall, specifically between mowing and burning, but there are indications as to the potential importance of water tables, temperature and sedge cover.

In summary, the measured and modelled gas fluxes described in Sections 4.2.14 and 4.2.15 show that:

- As expected, measured NEE under both full light and in the dark was overall greater in the uncut than mown or burnt plots after management intervention.
- After management, the burnt plots, unlike the mown plots, showed a measured net efflux of CO_2 at Nidderdale and Whitendale, but not at Mossdale. The measured CO_2 efflux in the dark was also lower on burnt plots.
- Up-scaled annual estimates of NEE showed that both burnt and mown plots switched from a net C sink to a net C source after management.
- Annual C losses were greater from burnt than mown plots in 2013, but 4 years after management, C losses from burnt and mown plots, averaged across the three sites, were very similar.
- On the uncut plots, the mean up-scaled NEE values over five years showed the wettest site (Mossdale) to be a net C sink, while the driest site (Nidderdale) was a small net C source.
- There was very high variability in the measured methane fluxes, which constrained environmental interpretation.
- There was no significant effect of management intervention on methane fluxes from non-vegetated areas.
- However, methane fluxes from vegetated areas were higher from uncut plots than from burnt plots.
- Methane fluxes were higher on Mossdale, the wettest site, and fluxes were positively correlated with the WTD in the four weeks prior to measurement.
- There was also a weak positive relationship between methane fluxes and soil temperature and sedge cover.
- Fluxes of N_2O at the three sites were consistently very low and showed no effect of management.

4.3 Up-scaling carbon budgets, net GHG emissions and peat accumulation

Whilst *Calluna* is found in many parts of oceanic Europe, about 75% of the world's *Calluna*-dominated moorlands (Tallis et al., 1998) and around 15% of the world's blanket bogs (Bain et al., 2011) occur in Britain and this globally rare habitat is thus considered of international importance (Thompson et al., 1995). However, although their ecological function in relation to carbon, water and biodiversity is evident and frequently mentioned (e.g. Bain et al. 2011), actual numbers on carbon storage, greenhouse gas emissions and stream export of carbon as both DOC and POC from well-defined catchments are still very limited. Most importantly however, there are hardly any studies available measuring all these components over the same time period and within the same area. This study addressed this knowledge gap to capture the necessary data for upscaling in space and time.

4.3.1 *Predicted carbon flux balance and carbon budgets*

The measured and up-scaled annual net ecosystem exchange (NEE) carbon CO_2 fluxes (Section 4.2.14) based on chamber measurements for the main managements allowed calculation of the cumulative net CO_2 flux balance over the four years after management intervention, which allowed a comparison to the uncut management over the same time period (**Table 17**). Overall, averaging data across the three sites, the uncut management sequestered carbon; in contrast, both mowing and burning lost similar amounts of carbon (about 400 g C m^{-2}). Importantly, the estimate for the burnt management increased considerably (to around 1,000 g C m^{-2}) when including the estimated loss of burnt carbon from standing heather (around 550 g C m^{-2}; see Section 4.2.5.2). However, in the long-term, losses from the decomposing brash layer would also need to be included in the cumulative C flux balance for mowing. Although Nidderdale showed the lowest carbon gains for uncut management (indeed, the values were slightly positive), losses from burnt and mown management were also lowest at this site. However, both Nidderdale and Whitendale showed a steeper downward trend in NEE flux (CO_2 uptake) on burnt compared to mown plots since the onset of management (see **Figure 86** in Section 4.2.14) indicating a possible earlier change from a net C-source to a net C-sink for burnt plots.

Table 17 Cumulative net carbon flux balance (g C m^{-2}) for the three sites for the main management options (uncut, mown, burnt) during the four years after management intervention (2013-2016) based on up-scaled CO_2 fluxes (see Section 4.2.14). Negative values indicate a net C gain (i.e. a C sink), while positive values represent a net C loss. Totals for the burnt treatment are provided both with (Overall burning C loss) and without (Burnt) the estimated loss (Burnt biomass C) from burning the standing heather biomass (determined by manual harvest, see Section 4.2.5.2). Note that the potential long-term decomposition C losses from the mown brash layer are not included in the cumulative estimate for mowing.

(CO2 flux only)	Cumulative C balance (gC m-2) ± SE		
2013-16	**Nidd**	**Moss**	**Whit**
Uncut	**96±30**	**-889±81**	**-445±56**
Mown	**333±27**	**446±7**	**520±26**
Burnt	**226±10**	**578±34**	**505±17**
Burnt biomass C	558	593	543
Overall burning C loss	**784**	**1170**	**1047**

However, in order to obtain a meaningful net rate of carbon accumulation it is important to also include other carbon fluxes in addition to NEE CO_2 fluxes. Other considerable losses occur from methane emissions, and also via fluvial losses as DOC and POC. The net ecosystem carbon balance (NECB) is the annual net rate of C accumulation for an ecosystem including such losses, with negative values indicating C gain and positive values indicating C loss (Chapin et al., 2006).

Compilation of the annual NEE (see **Figure 85** and **86** in Section 4.2.14), methane (**Table 16** in Section 4.2.15) and fluvial total DOC and POC (**Table 13** in Section 4.2.11) carbon fluxes, proportionally averaged across managements (see Net Ecosystem Carbon Balance (NECB) Section in Appendix 6), allowed NECB values for the uncut comparison areas to be estimated for each site and year (**Table 18**). For example, in 2013, ~20% of each sub-catchment was

mown and burnt respectively, leaving 80% of the entire site unmanaged, with another ~20% being managed in 2015, resulting in only 60% unmanaged area (but this was equal to about 50% of heather dominated areas) at each site. When averaged over the five years based on mean C flux components, the NECB values at all three sites were positive, indicating a net loss of carbon. However, results for individual years varied considerably for all sites, and particularly at Mossdale and Whitendale. Nidderdale consistently acted as a net source of C, although the amount lost varied year on year, with about seven times as much C lost in 2016 as in 2013. Mossdale acted as a weak C sink in 2012, a very strong C sink in 2013, and to a lesser extent in 2014, but became a strong C source in 2015 and more so in 2016, losing more C than was gained over the two previous years. Whitendale lost C in two years and gained C in the two other years, including in 2015 when both Nidderdale and Mossdale showed a net loss of C.

Table 18 The net ecosystem C balance (NECB) in each year for each site (either as the sum of mean or median values for the main C-flux components). Mean net ecosystem exchange (NEE), CH_4 fluxes, dissolved organic C (DOC_{1-2}; including all DOC fractions, see **Table 13**) and particulate organic C (POC) export (2012 for uncut (DN) management and 2013-2016 proportional averages of the uncut, burnt and mown catchment) were up-scaled (based on monthly flow volumes) to yearly site values (g C m^{-2} y^{-1}) and summed to derive NECB. The longer-term 5-year means are shown with the standard error (SE) over the five years. Grey shaded areas indicate entirely uncut pre-management values for each site.

Site	Year	NEE	CH_4	DOC_{1+2}	POC	NECB
Nidderdale	2012	-26.5	0.1	48.8	9.6	32.0
	2013	0.6	1.0	15.6	1.8	19.0
	2014	-10.1	8.3	30.5	2.4	31.1
	2015	64.3	5.3	29.6	2.5	101.7
	2016	67.9	51.8	20.6	1.5	141.9
	5-year mean	**19.2**	**13.3**	**29.0**	**3.6**	**65.1**
	SE	19.6	9.7	5.7	1.5	24.1
	Median	**0.6**	**5.3**	**29.6**	**2.4**	**37.8**
Mossdale	2012	-97.8	15.7	45.3	4.7	-32.1
	2013	-317.1	5.8	22.7	0.7	-287.9
	2014	-187.1	36.7	35.1	0.7	-114.5
	2015	39.7	106.5	45.1	1.4	192.7
	2016	-49.7	373.8	35.5	0.8	360.4
	5-year mean	**-122.4**	**107.7**	**36.7**	**1.7**	**23.7**
	SE	60.9	68.8	4.2	0.8	59.7
	Median	**-97.8**	**36.7**	**35.5**	**0.8**	**-24.8**
Whitendale	2012	-19.1	3.7	39.4	7.2	31.2
	2013	-86.6	1.1	12.7	1.8	-71.1
	2014	53.7	3.4	29.2	1.9	88.3
	2015	-100.7	19.0	30.2	2.7	-48.8
	2016	-16.2	30.0	24.3	1.7	39.8
	5-year mean	**-33.8**	**11.4**	**27.2**	**3.1**	**7.9**
	SE	27.8	5.6	4.4	1.1	68.9
	Median	**-19.1**	**3.7**	**29.2**	**1.9**	**15.7**

The NEE fluxes at Mossdale and Whitendale consistently showed a net C gain, apart from one year, whereas Nidderdale only had two years of net NEE C gain, with an average overall NEE C loss (**Table 18**). On average, the largest C losses at Nidderdale and Whitendale was via DOC export in stream water, although at Mossdale C loss from CH_4 emissions was greater than that from DOC export. On average, POC represented the smallest loss of C at all sites, with a value of around 13% of the DOC loss. However, the high NECB losses were based on mean fluxes containing very variable components such as methane fluxes and NECB losses were lower across two sites when based on the median values (a more robust measure of central tendency) of each flux component. In fact, NECBs based on the median fluxes related to the order observed with the NEE fluxes, with Mossdale turning into a considerable C sink of -24.8 g C m^{-2} y^{-1} and Nidderdale loosing half the carbon (37.8 g C m^{-2} y^{-1}). Moreover, the stream DOC and POC components might overestimate losses as they included steeper and eroding slope areas unrelated to the monitoring plots on flat to low slope areas.

From the component C fluxes, it was possible to estimate the NECB values, averaged over all sites, for the three management regimes (**Table 19**). Interestingly, all three management groups acted as net carbon sources in 2012, before the onset of management. In the subsequent four years post-management, the NECB values showed that the burnt and mown (with brash left) consistently acted as C sources, whereas the unmanaged areas varied between years in acting as C sinks or C sources, showing much greater variation than the burnt or mown areas.

Table 19 The net ecosystem C balance (NECB) averaged across the three sites (either as the sum of mean or median values for the main C-flux components) for burnt, mown (brash left) and uncut areas for 2012-2016. Net ecosystem exchange (NEE) and CH_4 fluxes, dissolved organic C (DOC_{1-2}; including all DOC fractions, see **Table 13**) and particulate organic C (POC) export for the sites (averages of burnt and mown catchments) were up-scaled (based on monthly flow volumes) to yearly site values (g C m^{-2} y^{-1}) and summed to derive NECB. All NECB components were up-scaled across their respective management to the entire catchment area (i.e. 100%) apart from the DOC and POC components for unmanaged areas, which are the averages of stream flow values from burnt and mown catchments. The longer-term 5-year means are shown with the standard error (SE) over the five years. Grey shaded areas indicate the pre-management (uncut) values for burnt and mown managements.

Site	Year	NEE	CH_4	DOC_{1+2}	POC	NECB
Burnt (FI)	2012	-31.7	2.9	42.4	7.4	20.9
	2013	143.5	1.3	17.5	1.2	163.6
	2014	126.1	20.3	29.4	1.6	177.3
	2015	74.7	13.6	35.0	1.8	125.1
	2016	91.7	92.8	26.9	1.1	212.4
	5-year mean	**80.8**	**26.2**	**30.2**	**2.6**	**139.9**
	SE	30.7	17.0	4.1	1.2	32.9
	Median	**91.7**	**13.6**	**29.4**	**1.6**	**136.3**
Mown (LB)	2012	-18.4	1.6	46.7	7.0	36.8
	2013	79.8	1.2	16.5	1.7	99.2
	2014	139.0	26.3	33.8	1.7	200.8
	2015	129.4	18.0	35.0	2.6	184.9
	2016	85.1	18.7	26.7	1.5	132.1
	5-year mean	**83.0**	**13.1**	**31.7**	**2.9**	**130.7**
	SE	27.9	5.0	5.0	1.0	29.7
	Median	**85.1**	**18.0**	**33.8**	**1.7**	**138.6**
Uncut (DN)	2012	-47.8	6.5	44.5	7.2	10.4
	2013	-195.9	2.9	17.0	1.5	-174.5
	2014	-92.9	14.4	31.6	1.7	-45.2
	2015	-66.2	62.2	35.0	2.2	33.1
	2016	-57.8	215.9	26.8	1.3	186.3
	5-year mean	**-92.1**	**60.4**	**31.0**	**2.8**	**2.0**
	SE	27.0	40.3	4.5	1.1	58.5
	Median	**-66.2**	**14.4**	**31.6**	**1.7**	**-18.6**

In the pre-management year, the annual NEE flux of all uncut and to be managed areas was negative (**Table 19**), indicating that vegetation carbon uptake outweighed losses from respiration. This remained the case in all years for the unmanaged areas, whereas the NEE measurements on burnt and mown areas showed a net C loss in the post-management years. The burnt NEE flux losses were highest in 2013, the year in which burning took place, and decreased in the subsequent two years, whereas mown NEE fluxes were a lower in 2013 than in 2014 and 2015, possibly reflecting a lag time in decomposition of brash. Overall, the largest return of carbon to the atmosphere from unmanaged areas was via CH_4 emissions, but was via NEE for burnt and mown areas. There were few differences in DOC export between treatments after management, but mean CH_4 losses were higher in uncut plots in 2015 and 2016. However, the impact of this difference was lost when median fluxes were used. On average, POC export represented the smallest C loss for all managements. Overall, the NECB based on the more robust measure of median flux components showed nearly identical NECB losses of around 137 g C m^{-2} y^{-1} for the burnt and mown management, whereas the uncut management showed a C gain of -19 g C m^{-2} y^{-1}.

The site NECBs (**Table 18**) indicated that, averaged over the five years of the study, averaged areas at all sites were a net carbon source when mean flux components were used but not when based on medians. There was a considerable inter-annual variation, which could have resulted in a very different conclusion, specifically in relation to the data used by the IUCN UK's Peatland Code, over a shorter study period. For example, when excluding methane emission results for 2016 calculated mean NECB values would have resulted in Mossdale becoming a considerable C sink, Nidderdale remaining a C source and Whitendale becoming a very small C sink. However, only in 2016 did all three sites act as a net C source; while Nidderdale was always a C source, Mossdale was a net C sink in 2012, 2013 and 2014, and Whitendale was a net C sink in 2013 and 2015. It is interesting that the three sites, which were all superficially similar at the start of the study and were all managed in the same way throughout the study, should produce three very different outcomes. Given that all three sites are located in north-west England and are therefore subject to broadly similar climatic conditions, and that all three sites are *Calluna* dominated, the fact that NECB value can be either positive or negative for different sites in the same year suggests that there are likely to be unmeasured, or even unknown, factors influencing the C dynamics on these peatlands. Key aspects in this respect are differences in vegetation composition and site wetness in relation to bog habitat condition, which are also discussed in Appendix 3a.

Annual variability in the NECB has been recorded in other studies. Measurements of the long-term NECB are relatively rare, although estimates for unburnt peatlands in Canada (-22 g C m^{-2} y^{-1}; Roulet et al., 2007) and Sweden (-24 g C m^{-2} y^{-1}; Nilsson et al., 2008) were very similar to the overall uncut median NECB values (**Table 19**). Values for peatlands in the UK are more variable; although a longer-term estimate (combining various data) for the same Scottish site gave similar values to the Canadian and Swedish NECBs (-28 g C m^{-2} y^{-1}; Helfter et al., 2015), other measurements report higher (-72.4 g C m^{-2} y^{-1}; Dinsmore et al., 2010) and lower (+8.3 g C m^{-2} y^{-1}; Billett et al., 2004) values of NECB. In comparison, the few studies which have specifically investigated rotationally burnt blanket bogs in the UK have all demonstrated a net loss of C, with Ward et al. (2007) showing a mean loss of 25.5 g C m^{-2} y^{-1} at Moor House, whilst Clay et al. (2010b) showed losses of 117.8 g C m^{-2} y^{-1} at the same site. Clay et al. (2015) demonstrated a range of losses of between 4 g C m^{-2} y^{-1} and 269 g C m^{-2} y^{-1} for areas of different burn ages at Moor House, by using age markers in the peat and quantifying the C store above this. Also, areas on a 10 year burn cycle were estimated to show an average reduced sequestration rate of 73 g C m^{-2} y^{-1} after three burn cycles compared to the unburnt areas on the same site (Garnett et al., 2000). However, continued longer-term flux monitoring, over a full management rotation (e.g. 10-15 years), is clearly needed to reach any meaningful conclusions about the C balance from mowing and burning treatments, particularly in relation to identifying a possible optimum management rotation length (i.e. the duration after which the C-sink status likely declines).

Crucially, none of the available literature report measured NECBs for more than one site over the same period using the same methods, meaning that much of the inter-annual variability in previous literature was attributed

to changes in the climate whereas, in this study, other factors, in addition to climate (which, for instance, is likely to explain the large increase in methane emissions in 2015 and 2016) appeared to cause variation in NECB. One potential reason for the NEE balance for the Mossdale site in 2015 (**Table 18**) showing a small C loss (rather than NEE net C gains as in all other years) could be due to damage of the *Calluna* plants by either heather beetle (*Lochmaea suturalis* (Thomson)) or *Phytophthora* species. It was noted that substantial areas of *Calluna* were browned or reddened and there is evidence from other work showing *Calluna* dieback following infestation by either heather beetle (Scandrett & Gimingham, 1991) or *Phytophthora* species (Orlikowski et al., 2004), thus decreasing photosynthesis and CO_2 uptake. However, it was not determined conclusively whether heather beetles or *Phytophthora* species were responsible for this discolouration of *Calluna*, nor whether this was the specific cause of the reduction in C uptake during NEE. Given the net C uptake via NEE was over 10 times lower at Mossdale in 2015 compared to 2013 (**Table 18**), more research into the causes of NEE variability and the impacts of natural pests such as heather beetle and *Phytophthora* damage is needed.

In contrast to most other studies, DOC export did not always represent the largest C loss at all sites in all years. This is because some of the CH_4 fluxes recorded in this study were an order of magnitude higher than those recorded or calculated in C budgets in other studies. In addition to the previous comparison to literature values (see Section 4.2.15), Dinsmore et al. (2010) calculated C losses via CH_4 to be less than 0.5 g C m^{-2} y^{-1}, although they did acknowledge that these values were low for a UK peatland. However, higher values of 2.7 mg CH_4 m^{-2} h^{-1} measured in Caithness using an eddy co-variance system (Hargreaves & Fowler, 1998), and of 3.5 mg CH_4 m^{-2} h^{-1} measured at Moor House (Ward et al., 2007), are still at the lower end of those recorded in this study. Although average 'hotspot' CH_4 fluxes of around 10 g CH_4 m^{-2} y^{-1} from ditches in Wales (Cooper et al., 2014) and mean annual bog emissions of up to 60 g CH_4 m^{-2} y^{-1} in a review by Abdalla et al. (2016) are similar to the 2015 averages at uncut plots at Nidderdale and Whitendale (**Table 16**) and mean site fluxes overall (**Table 18**). However, overall median values (as a more robust measure of central tendency as shown in Abdalla et al.'s (2016) review on bog and fen methane emissions) much reduced CH_4 carbon losses (**Table 18** and **Table 19**).

The size of the NCEB estimates of net C loss at the different sites including proportional management (**Table 18**) contrast to the observed peat depth (**Figure 12**), which increased from Mossdale (121 cm) to Nidderdale (155 cm) to Whitendale (168 cm). However, the NECB values also reflect fluvial DOC and POC losses. As fluvial losses (**Table 18**) also include losses from other areas and slopes with much shallower peat, different vegetation and likely higher runoff and erosion, this might have caused over-estimating fluvial losses in relation to plot-level measurements, therefore limiting linking catchment-scale fluvial losses to plot-level fluxes and peat depth. Therefore, only considering C-fluxes from NEE and CH_4 might provide a more robust indication of long-term peat accumulation. Indeed, excluding the fluvial DOC and POC fluxes (as reported in **Table 19**), the overall average site mean and median NECBs across all sites showed a net carbon accumulation of -32 or -41.0 g C m^{-2} y^{-1}, respectively. This provides a similar mean as for other UK NECB estimates (+8.3 g C m^{-2} y^{-1} in Billett et al., 2004 and -72.4 g C m^{-2} y^{-1} in Dinsmore et al., 2010). Moreover, when excluding fluvial losses the five year means at the site level (**Table 18**) for Whitendale, which had the largest peat depths, also showed the largest accumulated C (-22.3 g C m^{-2} y^{-1}) whereas there were net C losses at Nidderdale (32.5 g C m^{-2} y^{-1}) and a small C gain at Mossdale (-14.7 g C m^{-2} y^{-1}), which had shallower peat depths. These data highlight the need for a long-term and catchment-scale assessment of NECBs, considering topography and vegetation impacts on overall C accumulation and loss rates. Both are necessary in order to provide robust data underpinning carbon offsetting schemes at the landscape scale.

The consistency with which uncut (DN) areas took up CO_2 via NEE compared to the consistency and magnitude with which the burnt and mown areas lost C as NEE fluxes over the same period after management intervention (**Table 17**), strongly suggests that the DN management was most beneficial in terms of the NECB. This is in direct contrast to the findings of Ward et al. (2007) who found Moor House plots which were burnt took up more C than unburnt areas. However, although the burnt areas in that study were on a 10 year burn rotation, comparable to

this study, the plots were measured nine years into the cycle, whereas this study currently only extends for four years into the management cycle. So far, the 5-year mean annual (±standard error; median) NECB for the uncut plots considering only NEE and CH_4 fluxes showed an overall C gain of -31.7 gC m^{-2} (±48.2; -55.7 gC m^{-2}), consisting of a positive (C loss) NECB for Nidderdale with 30.5 gC m^{-2} (±36.0; -24.5 gC m^{-2}) and negative (C gain) NECB for both Mossdale with -42.0 gC m^{-2} (±139.4; -82.1 gC m^{-2}) and Whitendale with -83.6 gC m^{-2} (±45.2; -60.7 gC m^{-2}). A study by Clay et al. (2015) investigated more recent burns but also showed burning appeared to increase C uptake, with young burn scars (1-6 years since burning) tending to have a negative NEE flux. Nonetheless, when other C losses were included, Clay et al. (2015) demonstrated that all ages of burn caused net C release, which agrees with this study. Continuation of flux measurements would provide much needed long-term data in order to develop regression models allowing to predict how the different management scenarios (i.e. burning and mowing) likely accumulate carbon over longer time scales (so far this is not possible as the direction/shape of long-term change is not captured by the short-term monitoring).

Unfortunately, it is not possible to compare the mown NECB to literature values as, to our knowledge, this is the first C budget calculated for UK peatlands under a rotational mowing regime. Notwithstanding the time limitations, based on the NECB values so far, it appears that mowing is not better than burning for the peatlands in terms of C sequestration. However, no measurements were taken during management implementation. For mowing, there should not have been any additional C loss to the usual fluxes as the brash was left (only BR plots had brash removed and these mowing management variants were not up-scaled). For burning, the majority of the standing plant biomass was pyrolysed, although a small amount of this remained as charcoal, and there may also have been some peat lost in the fires. The latter was not possible to quantify but, from measuring standing biomass on plots before burning (see **Table 17**), up to 587 (± 155) g C m^{-2} of *Calluna* was burnt, with only about 20 g C (or 5%) of that biomass turned into charcoal (see Section 4.4.4). Therefore, for a 10 year burn cycle, an additional C loss of ~57 g C m^{-2} y^{-1} should be added to the NECB for the burnt plots, highlighting the potential better C sequestration potential of mowing compared to burning when including losses during biomass combustion. However, when comparing C budgets between burning and mowing, the assessment of the mown management (i.e. LB) would need to consider continuation of C losses from the initial management intervention as the exposed brash layer continues to decompose over time.

The C fluxes presented above allowed comparison of the sites to those classes reported and used by the IUCN UK's Peatland Code (Smyth et al., 2015). Overall, the comparison (**Table 20**) was close to the Peatland Code values. However, whilst DOC fluxes compared well, the Peatland Code's POC assumption of zero POC fluxes for 'modified' bog would need to be revised. Moreover, there was a large difference in the methane emissions, resulting in a higher overall net emission factor than the "modified" Peatland Code category, mainly due to the wettest site (Mossdale). Notably, this reflected the much higher methane emissions recorded in 2015 and 2016, but could likely represent future warmer years with higher winter but lower summer rainfall predicted to occur more frequently under climate change scenarios. Clearly these findings underline the need for long-term monitoring in relation to both policy-evidence (Lindenmayer et al., 2017) and scientific impact (Hughes et al., 2017), particularly in relation to net GHG emissions and the concept of "a wetter bog is a better bog". Notably, up-scaling by empirical models relies on complex and as yet considerably uncertain relationships between carbon flux components and climatic variables. However, this study found relationships similar to those previously reported elsewhere, as was discussed in earlier sections.

Table 20 Comparison of the IUCN UK's Peatland Code emission values ($tCO_2eq\ ha^{-1}\ yr^{-1}$) by component, and the corresponding net emission factors, to the project sites (Nidderdale, Mossdale and Whitendale) ranging from less to more modified blanket bog and data (for the experimental management scenarios, including uncut and in 2013 and 2015 increasing areas of burnt and mown management as shown in **Table 34**); same CO_2 equivalent calculations were applied as in in Smyth et al. (2015; cf. Table 1).

Peatland Code Category	Statistics	CH_4	CO_2	N_2O	DOC	POC	Emission Factor
Pristine*	-	-	-	-	-	-	**Unknown**
Near Natural	Mean (±StE)	3.2(1.2)	-3.0(0.7)	0.0(0.0)	0.88	0	**1.08**
	Median	1.5	-2.3	0.0			
Modified	Mean (±StE)	1.0(0.6)	-0.1(2.3)	0.5(0.3)	1.14	0	**2.54**
	Median	0.2	0.1	0.5			
Drained	Mean (±StE)	2.0(0.8)	1.4(1.8)	0.0(0.0)	1.14	0	**4.54**
	Median	1.0	-0.9	0.0			
Actively Eroding	Mean (±StE)	0.8(0.4)	2.6(2.0)	0.0(0.0)	1.14	19.3	**23.84**
	Median	0.1	0.4	0.0			

Project sites Drained & modified	Statistics	CH_4	CO_2	N_2O	DOC	POC	Emission Factor
Mossdale ('less modified')	Mean (±StE)	35.9 ± 22.9	-4.5 ± 2.2	**0.12**	1.2 ± 0.1	0.03	32.78
	Median	**12.2**	**-3.6**		**1.2**	**0.02**	**9.96**
Whitendale ('intermediate')	Mean (±StE)	3.8 ± 1.9	-1.2 ± 1.0	**0.12**	**0.9 ± 0.1**	0.06	3.65
	Median	**1.2**	**-0.7**		**1.0**	**0.04**	**1.66**
Nidderdale ('more modified')	Mean (±StE)	4.4 ± 3.2	0.7 ± 0.7	**0.12**	1.0 ± 0.2	0.07	6.29
	Median	**1.8**	**0.0**		**1.0**	**0.05**	**2.93**

4.3.2 *Predicted carbon sink/source range and net GHG emissions*

The combination of the up-scaled, site-specific carbon fluxes (previous Section 4.3.1) and the estimated fluxes of the greenhouse gas N_2O (see Section 4.2.15) allowed calculation of the net GHG emissions (expressed in CO_2 equivalents) over a 100 year time frame for the corresponding gas global warming potentials (GWP_{100}) for each site and management scenario (**Table 21**).

Table 21 Average annual (p.a.) CO_2 balance and annual net greenhouse gas emissions (GHGs) during 2012-2016 for uncut areas and, during 2013-2016, for burnt and mown areas. The GHG emission values (in tCO_2-eq km^{-2} yr^{-1}) use the up-scaled measurements for CH_4 and best estimates for N_2O fluxes for the three sites Nidderdale (Nidd), Mossdale (Moss) and Whitendale (Whit). The median (of the annual values) was used for the methane fluxes. Conversion factors for CH_4 (x25) and N_2O (x298) are based on IPCC (2007) values for global warming potential of these gases over a 100 year time period (GWP_{100}). Values of CO_2-gC were converted (x 3.66) into CO_2 molar mass (g).

	Average fluxes, CO_2 equivalents and net GHGs p.a.		
2012-16 (Uncut)	**Nidd**	**Moss**	**Whit**
CO_2-gC (m^{-2} yr^{-1})	14	-197	-93
CO_2 g (m^{-2} yr^{-1})	**51**	**-724**	**-340**
CH_4 g C (m^{-2} yr^{-1})	4.0	36.7	3.7
CH_4 g in CO_2 eq (x 1.33 x25)	**133.3**	**1223.0**	**123.3**
N_2O g (m^{-2} yr^{-1})	0.04	0.04	0.04
N_2O g in CO_2 eq (x298)	**10.43**	**10.43**	**10.43**
net GHGs tCO_2eq per km^{-2} yr^{-1}	**194**	**510**	**-207**
2013-16 (Burnt)	**Nid**	**Moss**	**Whit**
CO_2-gC (m^{-2} yr^{-1})	**56**	**144**	**126**
CO_2 g (m^{-2} yr^{-1})	**207**	**529**	**462**
CH_4 g C (m^{-2} yr^{-1})	1.9	35.4	13.4
CH_4 g in CO_2 eq (x 1.33 x25)	**63.3**	**1179.7**	**446.6**
N_2O g (m^{-2} yr^{-1})	0.04	0.04	0.04
N_2O g in CO_2 eq (x298)	**10.43**	**10.43**	**10.43**
net GHGs tCO_2eq per km^{-2} yr^{-1}	**281**	**1720**	**919**
2013-16 (Mown)	**Nid**	**Moss**	**Whit**
CO_2-gC (m^{-2} yr^{-1})	**83**	**112**	**130**
CO_2 g (m^{-2} yr^{-1})	**305**	**409**	**477**
CH_4 g C (m^{-2} yr^{-1})	19.9	14.6	13.9
CH_4 g in CO_2 eq (x 1.33 x25)	**663.2**	**486.5**	**463.2**
N_2O g (m^{-2} yr^{-1})	0.04	0.04	0.04
N_2O g in CO_2 eq (x298)	**10.43**	**10.43**	**10.43**
net GHGs tCO_2eq per km^{-2} yr^{-1}	**979**	**906**	**951**

The overall impact from N_2O emissions of 0.04 g m^{-2} yr^{-1} was negligible, even when converting to CO_2 equivalents this amounted to only 10 tCO_2-eq km^{-2} yr^{-1}. The only comparable long-term N_2O emission data for a UK blanket bog with very similar vegetation (Sheppard et al., 2013) reported very similar average N_2O emissions of around 0.03 g N_2O m^{-2} yr^{-1}. In contrast, methane emissions (5-year average median across sites of ~15.2 g C m^{-2} yr^{-1}; see **Table 18**) were key in determining the net GHG emissions and were similar to values in recent reports for rewetted bogs (Vanselow-Algan et al., 2015) with methane contributions of around 20-50 tCO_2-eq km^{-2} yr^{-1}. In fact, methane emission based on median fluxes (**Table 18**) converted into CO_2 equivalents (accounting for molar mass and a GWP_{100} factor of 25) ranged from 123 tCO_2-eq km^{-2} yr^{-1} at Whitendale (uncut) to 1,223 tCO_2-eq km^{-2} yr^{-1} for Mossdale (uncut). For the uncut management scenario, the average net GHG emissions across all three sites was 166 tCO_2-eq km^{-2} yr^{-1} and only Whitendale revealed a negative net GHG emissions of -207 tCO_2-eq km^{-2} yr^{-1}, with Nidderdale showing a low positive potential GHG emission of 194 tCO_2-eq km^{-2} yr^{-1} and Mossdale a high net GHG emissions contribution of 510 tCO_2-eq km^{-2} yr^{-1} (**Table 21**).

The net GHG emissions of fluxes for the burnt and mown management (**Table 21**) were all positive (i.e. net contributor to global warming). Although again management overall (combined burnt and mown) was highest on Mossdale, Nidderdale showed the lowest net GHG emissions reflecting a decline in water table (**Figure 45**) across the three sites and thus less methane emissions (**Table 16**). Whereas there was a large difference in the mean net GHG emissions on burnt sites of 973 tCO_2-eq km^{-2} yr^{-1}, ranging from the highest net GHG emissions of 1,720 tCO_2-eq km^{-2} yr^{-1} at Mossdale to the lowest at Nidderdale (281 tCO_2-eq km^{-2} yr^{-1}), the mean net GHG emissions across the mown sites of 945 tCO_2-eq km^{-2} yr^{-1} was very similar to the mean of burnt sites but varied only by 73 tCO_2-eq km^{-2} yr^{-1}. However, these emissions did not include any contributions from the management intervention itself, either from fossil fuel usage or from burnt biomass. Moreover, the GHG emission values would have been much lower for Mossdale without the very high methane emissions in 2015 and 2016, which highlights the impact methane emissions can have at very wet sites. Neither DOC nor POC fluxes were included in the GHG emission estimates as the long-term fate (i.e. decomposition into CO_2) of either is still very uncertain. However, conversion factors were included in **Table 20** to allow direct comparison to the IUCN UK's Peatland Code values (Smyth et al., 2015). The results highlight the need for long-term monitoring to capture any 'abnormal' years in estimates used for up-scaling across habitats or management.

The calculated project C fluxes and net GHG emissions for uncut plots (**Table 21**) were graphically compared to those used by the IUCN UK's Peatland Code (**Figure 89**). Overall, the comparison supported the direction of the Peatland Code values for net carbon fluxes and net GHG emissions (i.e. on average negative for CO_2 and positive for CH_4 and net GHGs). The overall mean values of the three sites associate best with the Peatland Code's 'near natural' category. However, the ranges of the 5-year mean site values and their standard errors (SE) were considerable. The generally drier and more modified Nidderdale site was closely associated with the 'drained' category values for CO_2 and CH_4 fluxes and net GHG emissions. Values for Whitendale and Mossdale aligned more with the Peatland Code's category of 'near natural' bog but with noticeably higher methane and thus GHG emissions for Mossdale. In fact, the observed mean values and ranges (i.e. SE) of CH_4 and mean net GHG emissions for the very wet and less modified site Mossdale (cf. **Table 20**) were about 10 times higher than the Peatland Code's values for 'near natural' bog (even when based on the 5-year median CH_4 emissions). This highlights the need to obtain more data, specifically for wetter and 'near intact' sites, as there is increasing evidence from this study and recent (as yet unpublished) Moor House data (personal communication Rob Rose, CEH, UK) that methane emissions from such wet and less modified or near natural sites are potentially underestimated and moderately lower water tables (as at Whitendale) might prevent resulting in years of massive CH_4 emissions, particularly at wetter sites. However, the IUCN UK's Peatland Code values reflect a lack of published data, and error ranges may be particularly high for methane fluxes. Recent advances in *in situ* carbon flux measurements and long-term monitoring such as in this project offer a valuable opportunity to improve such estimates. Particularly, assessing possible mean water table thresholds in relation to preventing high net CH_4 and

thus positive net GHG emissions on less modified or near natural blanket bog should be a future research focus, also at other sites.

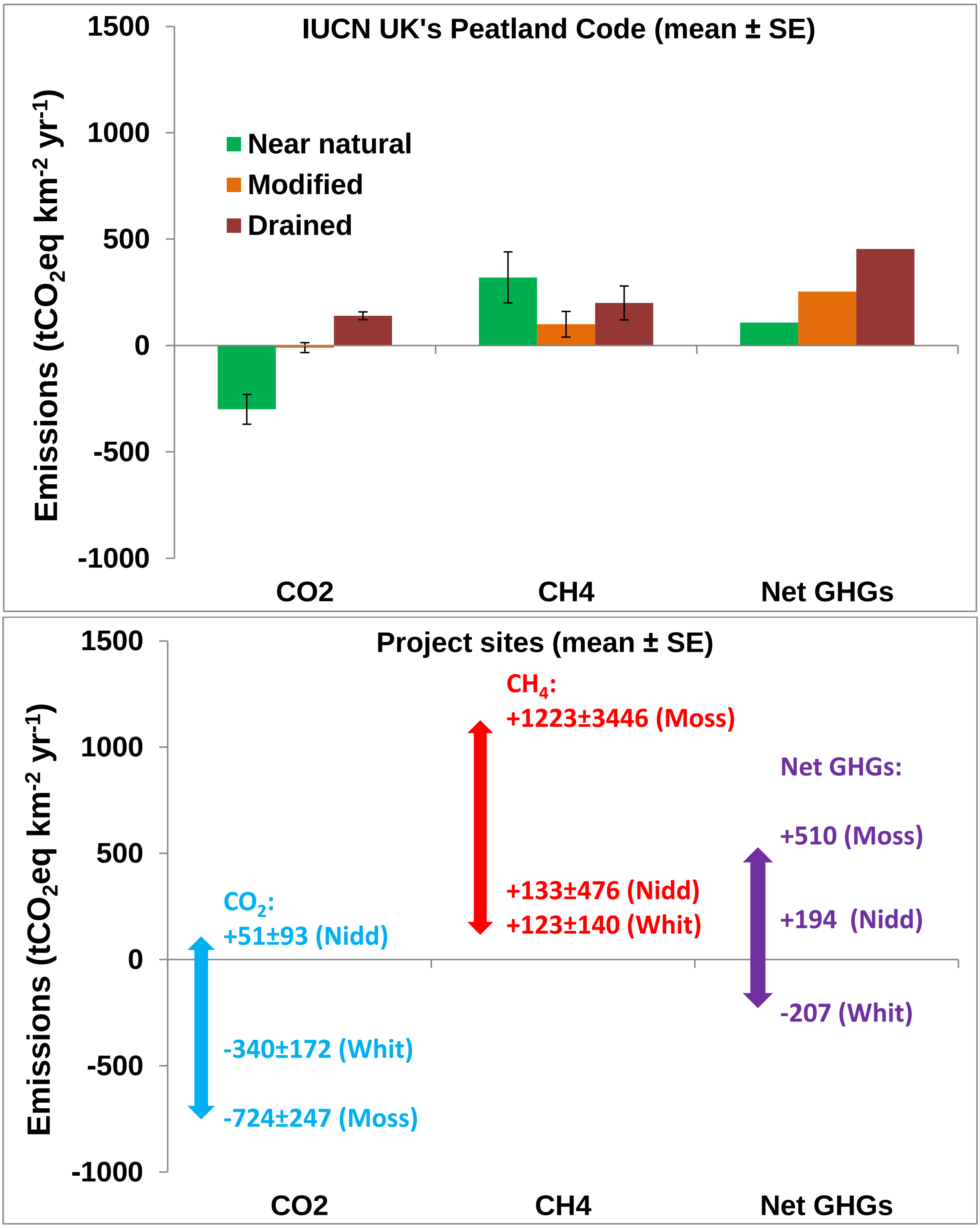


Fig. 89 Comparison of the IUCN UK (**top**) estimated mean C-fluxes (converted to $tCO_2eq\ km^{-2}\ yr^{-1}$) and net greenhouse gas emissions (net GHGs) calculated over a 100-year period as in the IPCC Fourth Assessment Report (2007) including CO_2-equivalents of CH_4 (GWP_{100} of 25) and N_2O (GWP_{100} of 298) emissions for the three blanket bog categories (i.e. 'near natural', 'modified' and 'drained') to the ranges (CO_2 shown as blue arrows, CH_4 shown as red arrows and purple arrows for net GHGs also including N_2O) of the three project sites (**bottom**) Nidderdale (Nidd), Mossdale (Moss) and Whitendale (Whit). The ranges (mean values are also shown in **Table 21**) are based on the sites' mean values (± standard error (SE) of the five annual budgets during 2012-2016) for the uncut plot-level management and are assumed to represent 'modified' blanket bog. Note: as the three shown IUCN UK's Peatland Code categories assume no POC export (cf. data in Smyth et al. 2015) measured POC export for the three project sites was also excluded.

In summary, the up-scaled estimates of NECB and net GHG emissions show that:

- Averaged over the five years of the study, uncut areas were either a very small carbon source (average NECB) or a small net carbon sink (median NECB).
- However, there was considerable variation between sites and years on uncut plots, and only in 2016 did all three sites act as a net C source.
- While annual NECB at Nidderdale was always a net C source, Mossdale was a net C sink in 2012, 2013 and 2014, while Whitendale was a net C sink in 2013 and 2015.
- The differences in NECB between sites and years were primarily due to variation in NEE, and to CH_4 fluxes being high in the wet years of 2015 and 2016.
- The estimates of both NECB and net GHG emissions for management scenarios and site averages were sensitive to the use of either mean or median CH_4 fluxes, particularly in 2016.
- After management intervention, median NECB values averaged across the three sites showed C losses under both mowing and burning that were on average 8 times larger than the C gain of the uncut scenario, primarily due NEE values of managed areas switching from negative (C gain) to positive (C loss).
- The size of the net C source in the burning scenario was higher than that of the mowing scenario, when the C loss during the burning itself was taken into account, but brash decomposition from mown plots would also need to be considered in the long-term.
- The use of fluvial C losses for NECB calculations, which included (likely greater) losses from steeper areas within the catchment with little or no peat cover, does not directly match CO_2 and CH_4 fluxes, which were measured predominantly on flat areas of deep peat.
- All mean net GHG emissions were positive, apart from the uncut management scenario at Whitendale.
- Mean net GHG emissions under the burnt and mown management scenarios were very similar (excluding additional emissions from either burning biomass or the management intervention); they were all positive and much greater than the uncut scenario.
- During the first three years of the study, the results agreed with current assumptions of the IUCN UK's Peatland Code, but the final two years resulted in very different values for net GHG emissions, particularly for the "least modified" (i.e. wettest) Mossdale site, mainly due to very high methane emissions.
- Long-term methane flux data highlight the need to gain more evidence in relation to possible water table and peat moisture thresholds for maintaining a net negative GHG emission balance on generally wetter and/or restored sites in relation to the view of a "wetter bog is a better bog".

4.3.3 *Past burn/fire frequency and peat carbon accumulation*

There is only sparse literature on the effects of burn management on actual carbon accumulation rates on blanket bogs (Davies et al., 2016). The lower decomposition rates and lower temperature sensitivities found on burnt plots in our study (Sections 4.2.13 and 4.4.2, respectively), and the lack of any meaningful differences in peat surface mean temperatures after burning (Section 4.2.12), suggest the potential for burning to alter net carbon accumulation through effects on decomposition rates. Moreover, the potential of charcoal to 'lock away' carbon over time (i.e. recalcitrant compound to decomposition) as suggested by Clay et al. (2010b) could explain the observed discrepancies in peatland carbon sequestration between flux and stock approaches as highlighted by Ratcliffe et al. (2017). In order to explore this further (in addition to the original aims), a peat core study was conducted in one burnt plot area at each of the three sites to relate carbon accumulation to past burn frequencies. We assessed two hypotheses:

1) Higher past burn frequencies correspond to larger long-term peat carbon accumulation.
2) Larger peat carbon accumulation shows a positive relationship with charcoal input.

At all three sites, two 1 m depth (5 x 5 cm diameter) peat cores were taken in early April 2016 from a burnt area next to a burnt plot (see **Figure 90**). Peat cores were transferred into square ducting (as shown in Section 4.4.1) and the top 25 cm of the peat profile was analysed for spheroidal carbonaceous particles (SCPs). These SCPs can be used as a dating tool (e.g. Swindles, 2011; see graph in **Figure 90**). Bulk density (BD), carbon content (C_{org}) and charcoal fragments were also quantified. Appendix 13 provides more detailed information on the methods. This allowed the effect of burn frequencies (determined by charcoal peaks in relation to the SCP age profile) on peat and carbon accumulation (estimated from the other variables) to be analysed.

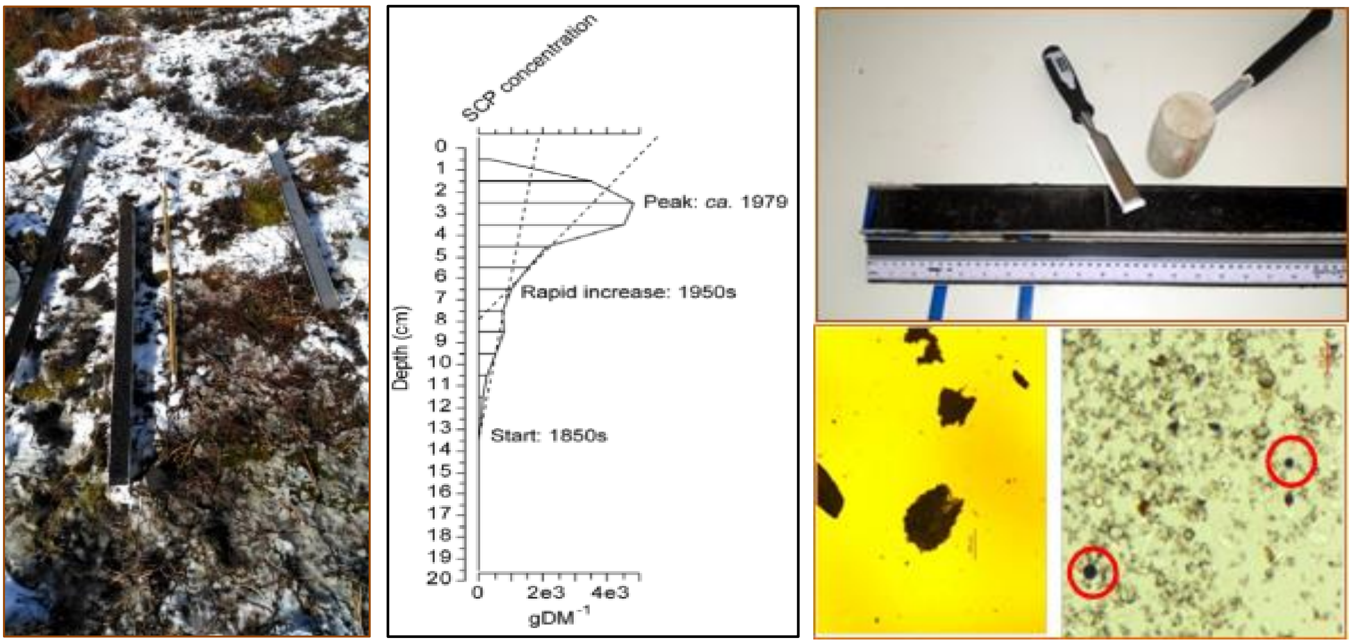


Fig. 90 Peat was cored to 1 m depth at all sites (**left**) and cohorts dated by the spheroidal carbonaceous particles (SCP) method (**middle**); ~2 cm^3 pieces were chiselled from the frozen core (**top right**) and were analysed for charcoal (**bottom right** fragments) to indicate past burn frequencies and for SCPs (**bottom right** red circles) to indicate carbon accumulation rates. Scale bars in the far right pictures are 200 µm for charcoal and 10 µm for SCPs. The expected depth-time profile for SCP concentrations over 20 cm (**middle**) is shown (reproduced from Swindles, 2010).

The patterns in BD, SCPs and C_{org} (from the two combined cores at each site) were similar for all three sites (**Figure 91**). There was a peak in BD of around 0.15-0.2 g cm^{-3} at a depth of about 5 cm, which coincided with the highest peak in SCPs at all sites. C_{org} did not show peaks and increased linearly with depth from about 53% to around 56%. However, the BD, SCP counts and C_{org} were lowest overall at Mossdale, and the SCP peaks at Mossdale and Whitendale were at 6 cm compared to 4 cm at Nidderdale. In agreement with Swindles (2010; **Figure 90**) SCPs could not be detected below 14.5 cm at Mossdale, 13 cm at Nidderdale and 14 cm at Whitendale.

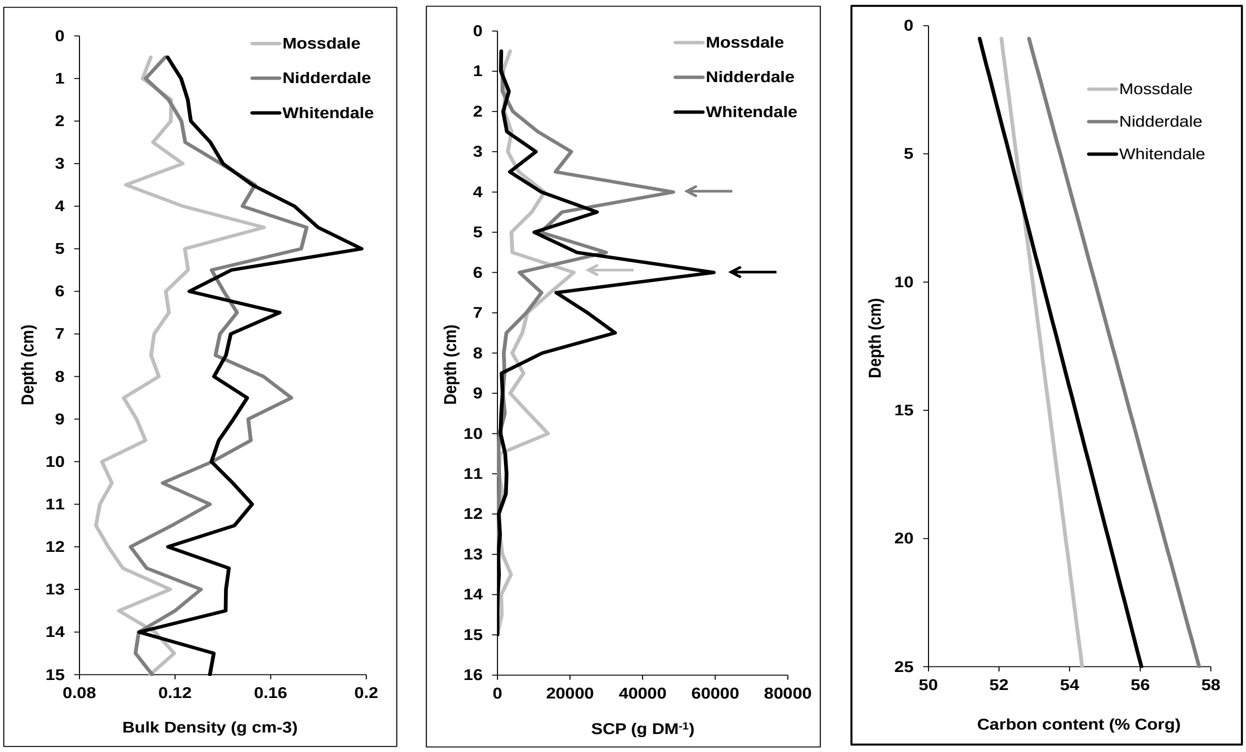
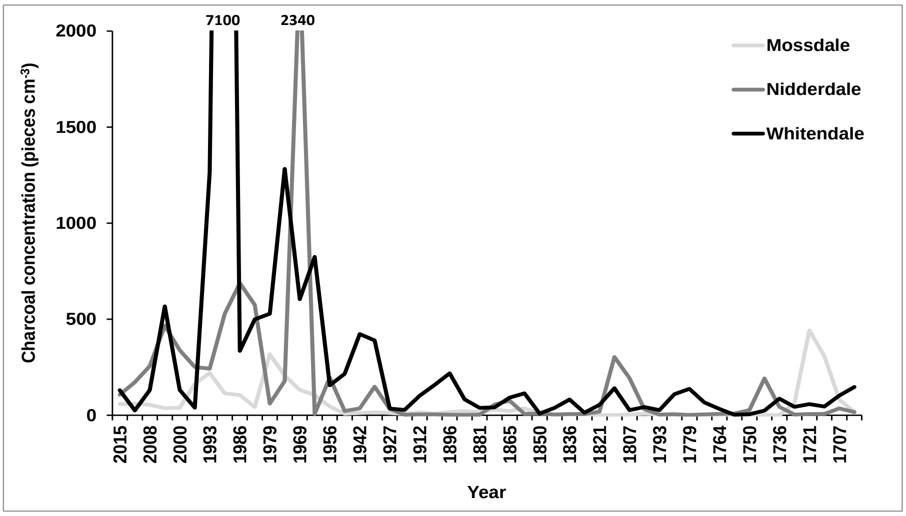


Fig. 91 Peat core depth profile for bulk density (**left**), SCP counts (**middle**) and % C_{org} (**right**) for each site. Bulk density and SCP counts were determined in 0.5 cm sections to a depth of 15 cm, whereas C_{org} was interpolated based on C/N analysis for 0-5 cm, 10-15 cm and 20-25 cm segments. Arrows indicate the peak SCP counts corresponding to 1975 (Swindles, 2010).

There were clear peaks in charcoal concentration throughout the peat profile. However, peaks were larger and more frequent nearer the surface (**Figure 92**; left). Together with the SCP-based dating tool, the charcoal peaks (size fraction >120 µm) provided estimates of past burn frequencies (**Figure 92**; right). Burn frequencies were shortest on average during the period 1950-2015 (every 17 years) and burns were most frequent, when averaged over the whole period since 1700 (depth of 25 cm), at Whitendale (every 23 years), less frequent at Nidderdale (every 25 years) and least frequent at Mossdale (every 28 years). However, the actual charcoal peaks were highest at Whitendale and Nidderdale, with very low counts for the less modified Mossdale site during 1850-1950; thus burn frequencies for Mossdale might be less in comparison to the other two sites than the 25 years indicated for the period 1850-1950 in **Figure 92** (right) as this was based on very low charcoal counts per peaks.



Burn frequency	1950 till 2015
Moss	22
Nidd	16
Whit	13
Burn frequency	**1850 till 1950**
Moss	25
Nidd	33
Whit	25
Burn frequency	**1700 till 1850**
Moss	38
Nidd	25
Whit	30

Fig. 92 Charcoal concentrations through the peat core depth profile for the three sites (Mossdale, Moss; Nidderdale, Nidd; Whitendale, Whit), determined for each 0.5 cm section to a depth of 25 cm but shown as the year each depth relates to (**left**). The y-axis is truncated to allow peak identification (values are given where high peaks are cut off). The resulting burn frequencies based on charcoal (>120 µm size) peak frequencies during specific periods (based on SCP dating) reflect average periods of management changes (i.e. onset of grouse management in 1850 and general intensification from 1950).

The resulting carbon accumulation rates (mean ± standard deviation), based on BD and C_{org} for the SCP dated sections (**Figure 93**), were significantly higher (82 ± 25 g C m^{-2} yr^{-1}) during the most recent period (1950-2015) compared to 1850-1950 (41 ± 5 g C m^{-2} yr^{-1}) and 1750-1850 (47 ± 2 g C m^{-2} yr^{-1}). Whilst the accumulation rates in the most recent period at Whitendale were only marginally significantly higher (p = 0.096) than at Mossdale or Nidderdale, rates were significantly higher at both Nidderdale and Whitendale than at Mossdale between 1850 and 1950 (p< 0.001) and higher at Nidderdale during the oldest period (p< 0.001).

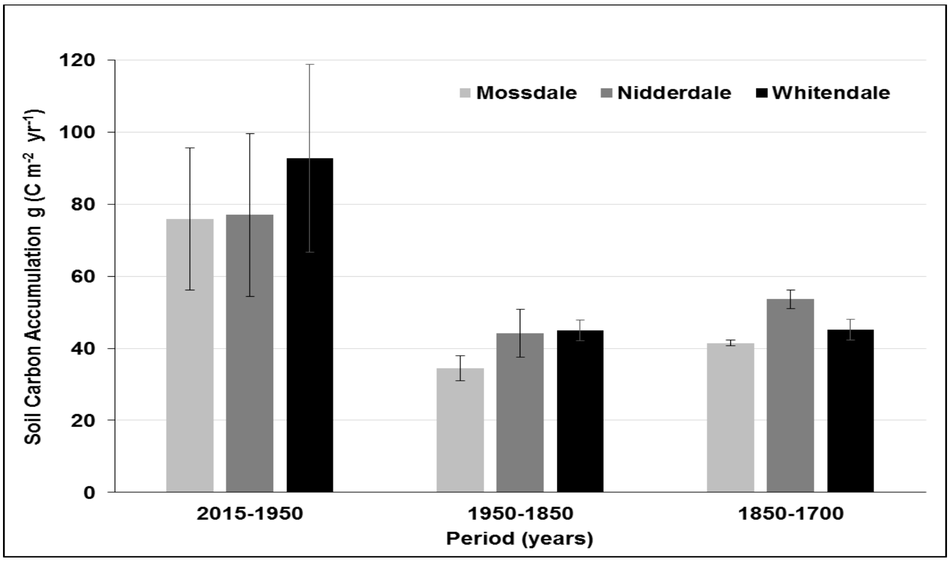


Fig. 93 Soil carbon accumulation rates, based on SCP dating the peat cores together with detailed bulk density and interpolated organic carbon content from up to five depth layers. Mean (± standard deviation) rates were calculated for each site for separate periods reflecting approximate times of management changes (i.e. onset of grouse moor management in 1850 and intensification from 1950). Accumulation rates differed significantly between sites in the latter two periods (2015-1950: p = 0.096; 1950-1850: p< 0.001; 1850-1700: p< 0.001).

Burning causes gaseous N-losses from combustion, which could have reduced N-availability in the soil organic matter (SOM) and may explain the observed increase with depth in peat C_{org} (**Figure 91**). Together with the increased BD, this resulted in the higher C accumulation rates obtained for the more frequently burnt sites over the three periods. In fact, mean C accumulation rates (2015-1950) of 3.0 t CO_2 ha^{-1} yr^{-1} (82 g C m^{-2} yr^{-1}) were very similar to the 3.8 t CO_2 ha^{-1} yr^{-1} as reported by Evans et al. (2014) for unburnt management based on data in Garnett et al. (2000). This was unexpected as, so far, the key study assessing burn rotation effects on peat carbon accumulation rates found a considerable carbon loss under burning over a similar time period (Garnett et al., 2000; see comparison **Table A13.2** in Appendix 13). However, Garnett et al. (2000) dried BD samples for only 24 hours, assumed C_{org} to have a constant value of 50%, and possibly used less accurate SCP methods. In addition, their reported charcoal layers did not agree with the oldest burn date (i.e. the onset of the experimental burn rotation in 1954 on all plots; cf. Figure 3 in Garnett et al., 2000). Notably, none of the depth profiles in Garnett et al. (2000) show the expected SCP peak around 1975, but all show a clear and high charcoal peak at about 10-11 cm depth. However, this was not considered in their age determination, although it most likely indicated the year 1954 (i.e. the last time all plots were burnt and the onset of the burn experiment). Together, these uncertainties mean that the peat C accumulation rates may have been more similar between burnt and unburnt plots than was suggested by Garnett et al. (2000). In fact, two other studies, Grand-Clement (2008) and Ward et al. (2007), on the same site as Garnett's study, which were not included in the review by Evans et al. (2014), showed higher peat accumulation rates on burnt than unburnt plots (Grand-Clement, 2008) or equally high C accumulation on burnt and unburnt plots over the top 1 m of peat (Ward et al., 2007) based on coarse sampling at 10 cm depth increments. The disagreement between burn frequencies and C accumulation in the mid period (1950-1850) in this study (**Figures 92** & **93**) could reflect methodological challenges; as charcoal concentrations were very low for Mossdale in older than 1850 layers (**Figure 92**), and depth layers are also closer together (thinner), so accurate

charcoal peak detection and separation as well as dating became less reliable. Ideally ^{14}C or other suitable dating techniques would be deployed to better resolve the age resolution.

A comparison of the peat C accumulation rates reported here to published data from other peatland sites over corresponding time periods (see **Table A13.2** in Appendix 13) revealed very good agreement, particularly when comparing this study to other blanket bog studies (Billett et al., 2010; Garnett, 1998; Hardie et al., 2007). Carbon accumulation rates in these studies are generally much higher during the most recent periods (about 50 - 100 g C m^{-2} yr^{-1}), reflecting highly undecomposed peat, whereas long-term accumulation rates for older layers are about 30 g C m^{-2} yr^{-1}. All three sites showed a long burn history, with the overall more frequently burnt and more "degraded" sites Nidderdale and Whitendale showing higher C accumulation (**Figure 93**), whereas the more "intact" site Mossdale showed less (although burn frequency was equally high for Mossdale and Nidderdale during 1850-1950; **Figure 92**). Three processes could explain this: (1) burning converted otherwise decomposable biomass carbon into 'inert' charcoal (~18 g m^{-2}, see Section 4.4.4), (2) the bulk density increased, possibly due to incorporation of ash and charcoal fragments thus increasing C stocks, and (3) a potential negative priming effect on decomposition by charcoal (see Section 4.4.2; Lu et al., 2014).

The chamber based carbon fluxes (see Section 4.2.14) showed that the highest C uptake was on Mossdale, the site that was the least modified (wettest) and least frequently burnt both since 1950 and overall since 1700. This is consistent with the concept that burning over time leads to a decline in peat C stocks (Garnett et al., 2000), but not with the considerable C accumulation shown in **Figure 93**. Hence, the conclusions reached here are different using the flux or stock change approach. Major disadvantages of the carbon flux approach are that it does not capture long-term incorporation of carbon as charcoal (Clay et al., 2010b), and that it captures decomposition from deeper, older layers, which then affects the carbon budget for recent carbon uptake, due to the mixed age of the decomposition signal. The major disadvantages of the carbon stock approach are that it relies on uncertain dating techniques (SCP) and considers sections of peat separately, which ignores incorporation of surface carbon into deeper sections through roots and changes in decomposition rates over time.

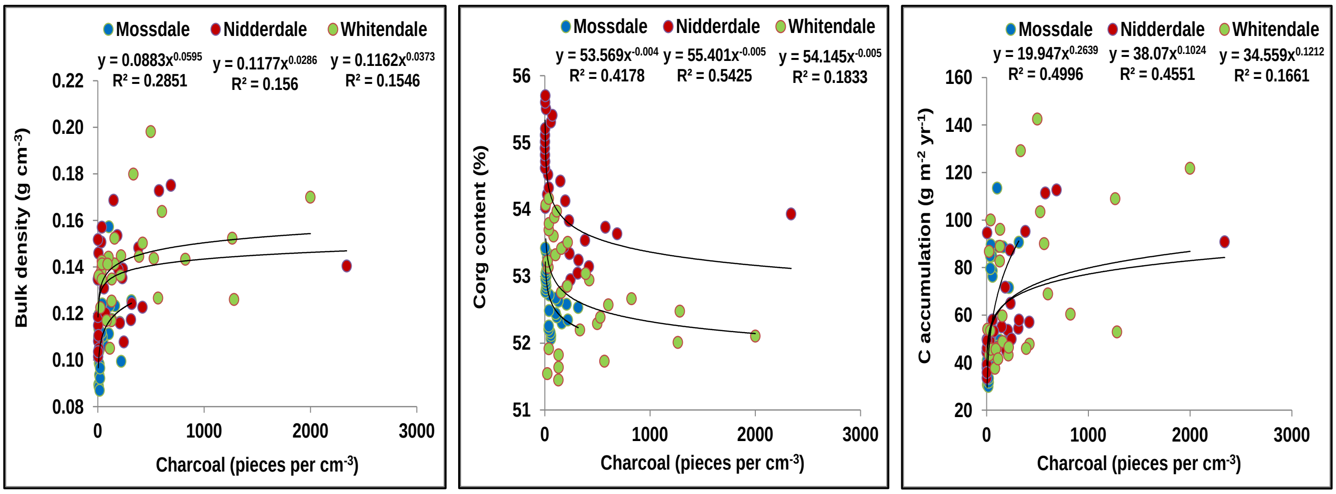


Fig. 94 Bulk density (**left**), organic carbon content (**middle**) and carbon accumulation (**right**) versus number of charcoal pieces per cm^3 of peat (concentration) from the top 15 cm (i.e. over the SCP and bulk density measurement range) of the three peat cores from Nidderdale, Mossdale and Whitendale. The best fit power functions are provided for each site.

Moreover, although burnt plots showed overall lower water tables (see Section 4.2.7) and thus should have increased decomposition rates, this was not the case. In fact, burnt plots showed reduced soil respiration fluxes (**Table 14**). If increasing bulk density through charcoal and ash incorporation decreases the available pore space, less water would be required to create anoxic conditions on burnt plots, as compared to the less dense and more open peat on unburnt plots. Indeed, the three sites revealed significant (see **Table A13.3** in Appendix 13) correlations between BD, C_{org}, carbon accumulation rates and charcoal concentrations (**Figure 94**, above).

Importantly, the greatest impact (greater BD and C_{org}) occurred on the overall most frequently burnt sites Nidderdale and Whitendale. However, whereas BD showed a positive relationship with charcoal piece abundance, C_{org} showed a negative relationship. Moreover, the R^2 values for the natural log transformed regressions were highest for Mossdale (see **Table A13.3** in Appendix 13), possibly relating to a more uniform distribution due to there being less charcoal (i.e. lowest counts) as the burns are less frequent and less intense.

This study supported the two hypotheses that (i) higher past burn frequencies correspond to larger long-term peat carbon accumulation, and (ii) that larger peat carbon accumulation rates positively correlate with higher charcoal amounts in corresponding peat layers. This highlights the potential of long-term charcoal C sequestration on rotationally burnt peatlands, possibly also supported by indirect impacts on decomposition due to changes in BD and possible impacts on microbial activity, and revealed discrepancies between the flux and stock approaches in peat carbon sequestration. However, SCP dating is uncertain and other factors may have also influenced these findings regarding charcoal impacts in relation to burn frequency (e.g. changes in N or vegetation composition and thus litter quality). Moreover, the negative relationship between C_{org} and charcoal abundance could be due to the interpolation of data from only three peat layers (see methods in Appendix 13); a more detailed C_{org} analysis should be included in further work, ideally extending this work to other sites across the UK. However, a subsequent more detailed BD and C_{org} analysis at 0.5 cm resolution throughout the peat profile has since shown more robust findings between BD, C_{org} and charcoal (Heinemeyer et al., 2018). Further research is also needed to assess the impact of increased charcoal and ash incorporation on peat hydrology and the potential eco-hydrological feedbacks on decomposition processes as highlighted by Ratcliffe et al. (2017).

In summary, the results from the modelled and up-scaled fluxes used to estimate the NECB suggested that, after C losses from the biomass burning process (but not losses from long-term brash decomposition) were taken into account, burnt plots were a greater net C source than mown plots (see Section 4.3.1). However, the analysis of actual peat C accumulation rates in cores from the three sites, using SCP dating, that was described in this section supported the two initial hypotheses and showed that:

- The depth profiles of SCP, BD and C_{org} content were similar at all three sites.
- C accumulation rates for all three field sites under burn rotation management were similar to the only previously reported estimate for unburnt management (in a burn comparison using similar methods) over the same period.
- Averaged over the three sites, burns were more frequent, and C accumulation rates were also higher, over the period since 1950 than in the period 1700-1950.
- Carbon accumulation rates over the periods 1950-2015 and 1700-1850 were greater on the most frequently burnt site.
- This was not consistent with the NECB estimates which showed the greatest net C losses from burning compared to either uncut or mown plots when including C losses during burning (not accounting for long-term losses from brash decomposition on mown plots).
- Whereas BD and C accumulation rates showed a positive (power function) relationship with charcoal piece abundance, C_{org} showed a negative relationship. However, the overall R^2 values of the regressions were low due to the high error in the measurement of any of these parameters and would benefit from additional research.
- The least modified site, Mossdale, with by far the most *Sphagnum* cover, showed the lowest burn frequency in recent history (since 1950), which could suggest that the greater modification of the other sites (i.e. vegetation Section 4.2.6) correlates with increased burning frequency/intensity (which could be investigated further by paleo-ecological studies of plant fossils).

- Importantly, our results do not allow a comparison to an unburnt scenario and estimates are based on low severity prescribed burns and the impacts of more severe arson or wildfire are likely to differ (i.e. when peat burning occurs).
- Finally, any holistic burn impact assessment should ideally be providing comprehensive assessments including above-ground, hydrological, gaseous and below-ground parameters in estimating catchment carbon stocks/fluxes (specifically considering the above outlined limitations in this study).

4.4 Assessments of peat level changes, decomposition processes, peat chemistry and mycorrhizal priming

4.4.1 *Peat shrinkage and expansion*

Peat surface level changes due to water table depth (WTD) or pressure changes are well known (Strack et al., 2006) and are sometimes referred to as "bog breathing" (Ingram, 1983). However, little is known about how this process is affected by different vegetation types and if these processes cause irreversible changes in the peat which might prevent it recovering to a 'natural' level after periods of drought. Moreover, so far it is unknown how changes in peat surface levels affect bulk densities (BD), particularly on blanket bogs. Importantly, as the most severe BD changes are likely to occur at the peat surface (where the greatest changes in soil moisture occur), such changes could significantly affect peat carbon stock calculations that focus on the surface layers. For example, Bellamy et al. (2005) found that English and Welsh soils with organic carbon contents (C_{org}) higher than 50 g kg^{-1} lost carbon in soil samples taken between 1978 and 2003 and the rate of loss increased with increasing $\%C_{org}$, and hence was particularly high on blanket bogs. However, Bellamy et al. (2005) only sampled the top 15 cm of soil, BD values were not measured (but instead derived from a generic equation; cf. Table 1 in Bellamy et al., 2005) and samples were unlikely taken at the same time of year. Therefore, if soil moisture was less and hence BD and C_{org} density was greater in the first sampling period, a wetter second sampling could have resulted in giving the appearance of reductions in soil C_{org} stocks over the same surface layer thickness. Additionally, vegetation can alter peat structure and possibly BD, and the vegetation growing on the areas sampled may have been different, as resampling was not done in exactly the same place and some areas had changes in vegetation and management over time (personal communication Guy Kirk, Cranfield, UK).

This study measured peat surface fluctuations under different vegetation types, managements and WTD changes to evaluate the extent to which physical impacts could have influenced the carbon stock changes reported by Bellamy et al. (2005) for highly organic soils. A combined approach of controlled laboratory experiments and field monitoring (see Appendix 9 for further methodological details) assessed two hypotheses:

1) the calculated 'apparent' changes in C_{org} densities can be explained by peat surface (depth) fluctuations, which cause significant changes in BD in the surface peat without any 'real' changes in C_{org} occurring, and
2) these fluctuations can be affected by land management, vegetation and WTD changes.

Laboratory experiment (see **Figure 95**; left):

In January 2014, eight one metre deep, 5 x 5 cm, peat cores (see Section 4.2.2) were taken from three vegetation types (*Sphagnum*, *Calluna* and *Eriophorum*) at Mossdale. The water table depth during sampling was at the surface. Cores had the first few centimetres trimmed off to remove the vegetation/litter layer, and were placed in open ended 1 m long square ducting pipes with one clip-on side to allow free movement of water. The 24 ducting pipes (8 per vegetation type) of identical diameter were placed in a glasshouse inside 10 cm diameter drainpipes with holes drilled at the target WTD; these were filled with water to the required level by using bungs to seal holes. All cores were allowed to settle at a medium WTD (-15 cm) for 4 months. The cores were then paired by vegetation type and each core in a pair was allocated a different WTD treatment (n = 4): wet (0 cm) versus dry (-35 cm) for 9 months, switching to dry (-35 cm) versus wet (0 cm) for 5 months, switching both to wet (0 cm) for 3 months, followed by switching to very dry (-100 cm) over the final two months. The initial WTD differences reflected a typical summer WTD drawdown and the switching assessed potential recovery rates over time. Surface level changes were measured using callipers and ruler. The top 5 cm of all cores were removed and measured for bulk density after the first 9 months of cores being wet (0 cm) and dry (-35 cm).

Fig. 95 Laboratory (**left**) and field (pair on **right**) assessment of peat shrinkage and expansion in relation to water table (moisture) changes. Peat cores (1 m deep; 5x5 cm) were taken at Mossdale and incubated in plastic tubes at various water levels (far **left**), similar to water table depths observed in the field. Bulk density changes were determined for top peat layers (second from **left**). Field data were collected using metal rods inserted into the bedrock with a flexible (loose) marker (far **right**) indicating the peat surface in addition to an incision mark on the actual rod (second from **right**).

The change in peat height over the experimental period, compared to the starting height of the glasshouse cores, is shown in **Figure 96** and ranged from -3.48 cm to +1.22 cm with a mean of -0.67 cm. Across all WTD changes, the greatest range of vertical movement in a single core was 3.57 cm and the smallest was 0.78 cm (both were *Eriophorum*-topped cores), with a mean of 1.99 cm across all cores. Notably, the biggest changes in peat surface for the dry treatments occurred over the first 3 months after cores were first allocated to the wet and dry treatments, and then remained fairly stable. Importantly, the first 9 month dry period in the dry treatment caused nearly irreversible shrinkage, with some recovery in *Sphagnum* cores but little recovery in *Calluna* and *Eriophorum* cores in the subsequent wet period; in contrast, the second 5 month dry period, which was experienced by the initial wet treatment, was followed by full recovery for *Sphagnum* and a partial recovery for *Calluna* and *Eriophorum* peat. This indicates the importance of *Sphagnum* for peat resilience against drought impacts. The final dry period (-100 cm) caused very fast shrinkage in both treatment sets.

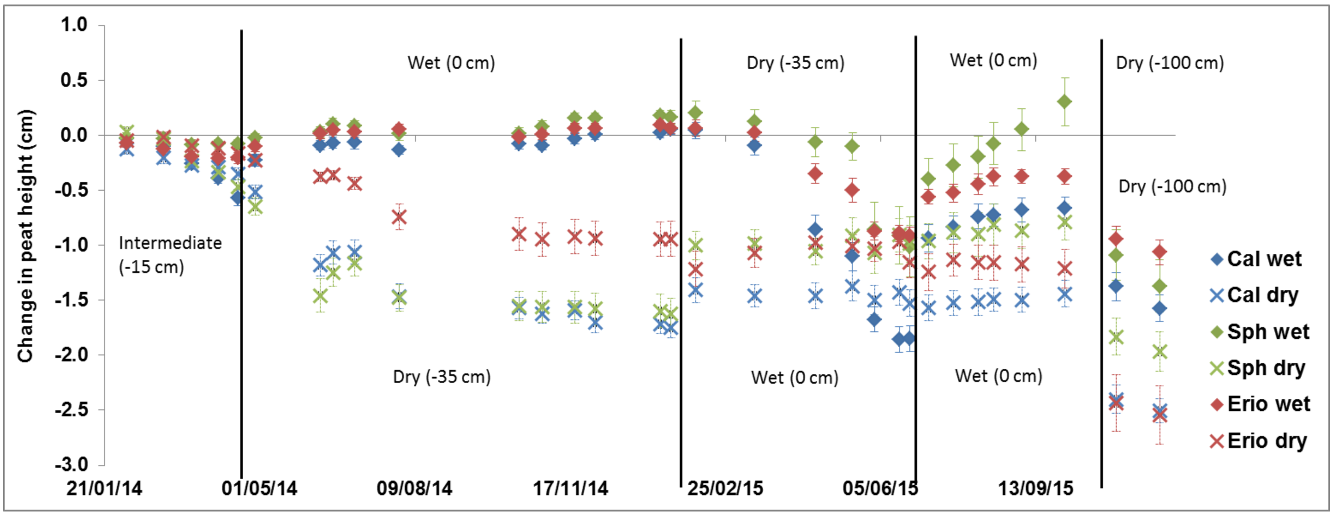


Fig. 96 Average (± standard error) shrinkage and expansion (compared to the initial peat surface) of the glasshouse peat cores under various water table depths (WTD) as indicated on the graph (cores were initially at -15 cm, then one half was wet (0 cm), the other dry (-35 cm), then treatment was switched, then all were wet and then all were dry). Infilled diamonds track the rise and fall of the peat surface for cores which were allocated to the wet (0 cm) treatment immediately following the initial intermediate period (-15 cm) and crosses track surface fluctuations of cores allocated to the dry treatment (-35 cm) during the same period. Times where WTDs were changed are indicated by black lines. Different colours indicate the different peat cores under predominantly *Calluna* (Cal) *Sphagnum* (Sph) or *Eriophorum* (Erio) surface cover.

The change in the height of the peat cores differed significantly between the three vegetation types ($F_{2, 90} = 8.26$, $p = 0.0005$), with *Calluna*-topped cores shrinking more on average than *Sphagnum*- or *Eriophorum*-topped cores ($p< 0.0003$) (**Figure 97**; top). However, there was no interaction between vegetation type and WTD treatments

($F_{8, 90}$ = 0.92 p = 0.50). When grouped into a "set" which was first allocated the wet treatment and a "set" which was first allocated the dry treatment, combining data for all vegetation types, the different sets showed a significant interaction with time periods ($F_{4, 90}$ = 8.39, p< 0.0001). The differences occurred when WTDs first diverged (wet cores gained 0.1 cm whereas dry core lost 1.4 cm), when all cores were wet (when initially dry cores were about 0.75 cm shorter than wet cores) and when all cores were dry (when initially dry cores were about 1 cm shorter than wet cores), with all cores being shorter than at the start, even during the period when all cores were wet (**Figure 97**; bottom), apart from the remaining wet cores.

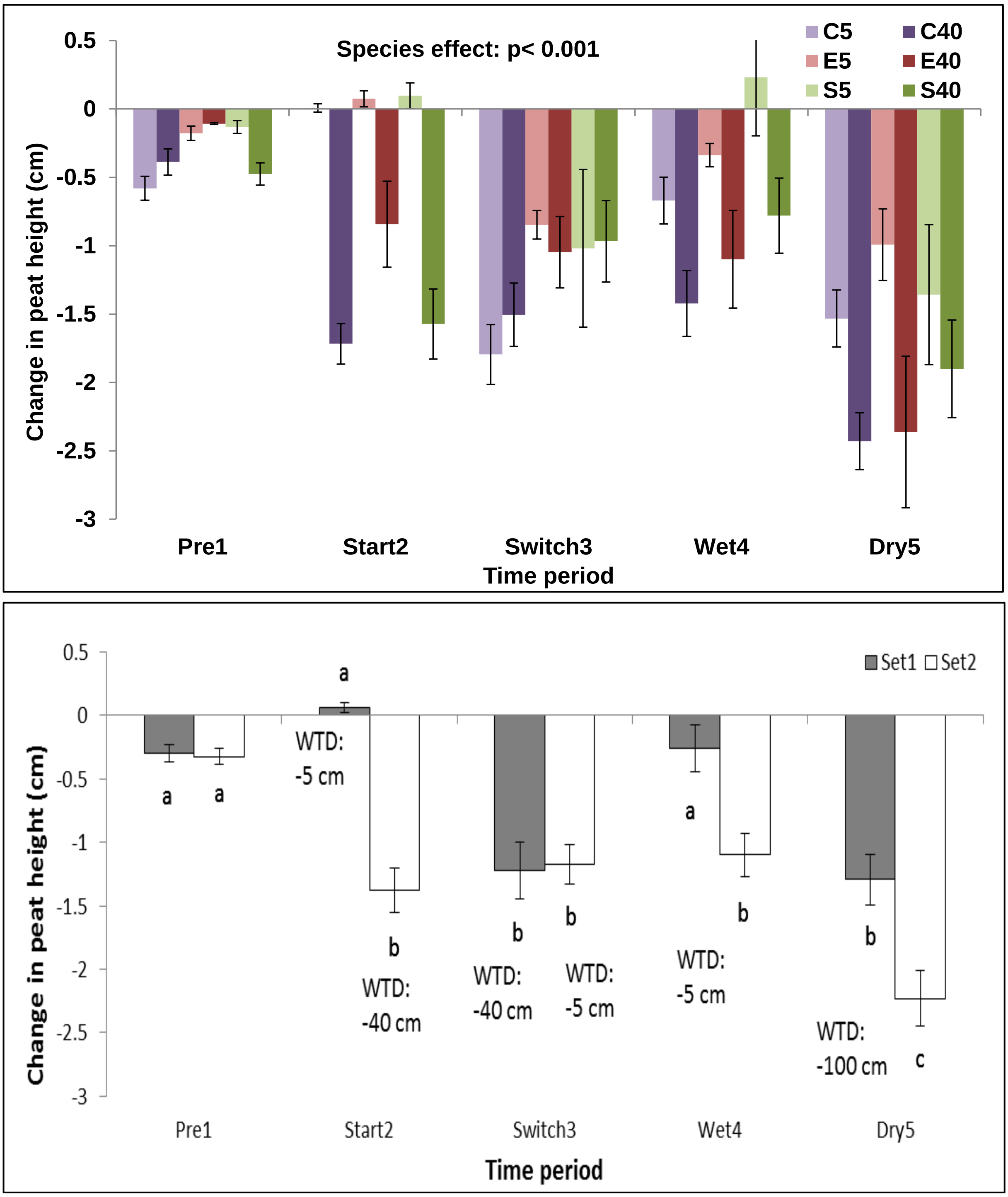


Fig. 97 Average (± standard error) shrinkage and expansion (compared to the initial peat surface) of the glasshouse peat cores under the five water table depth (WTD) periods: Pre1 (all -15 cm), Start2 (half at either -5 cm or -40 cm WTD), Switch3 (switched wet and dry), then Wet 4 (all cores wet) and Dry5 (all cores dry at -100 cm WTD). **Top**: individual treatments with peat cores under predominantly *Calluna* (C) *Sphagnum* (S) or *Eriophorum* (E) surface cover; **bottom**: combined cores for each WTD treatment. Significant differences (ANOVA) are shown by different letters.

When the bulk density (BD) was determined after the 9 month dry period in the laboratory experiment, values were higher in the dry cores for the *Calluna* and *Sphagnum* samples, but *Eriophorum* cores showed a non-significant decline in BD (**Figure 98**; top). There was a significant interaction between WTD and vegetation ($F_{2, 12}$ = 6.40, p = 0.0128), with the *Sphagnum*-dominated dry peat cores having a significantly higher BD than when wet or as dry *Eriophorum* cores (both p < 0.05). The lower BD in dry *Eriophorum* cores could be related to the very different nature of peat - whereas peat composed of *Calluna* or *Sphagnum* is horizontally layered and easily breakable, *Eriophorum* roots are fibrous and mostly vertical, creating strong and vertically structured peat, which is difficult to break and seems to prevent shrinkage. Moreover, *Eriophorum* roots can penetrate very deep, to at least 80 cm (Metsävainio, 1931), mainly functioning as a gas channel for transport of oxygen to deeper layers. However, this also allows upward water transport from deeper layers, which may have counteracted the drying.

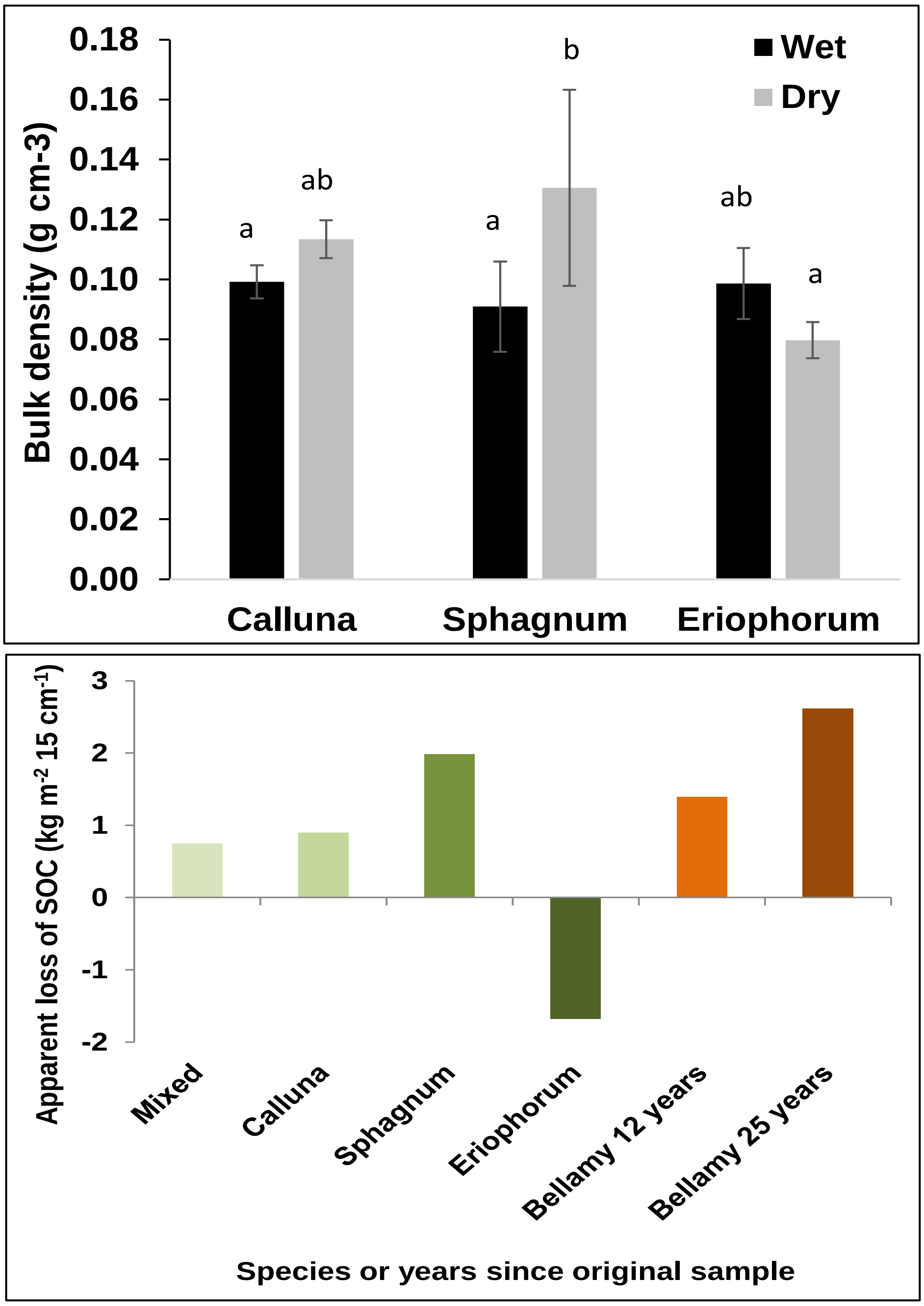


Fig. 98 Mean bulk densities (± 95% confidence interval) for wet (0 cm WTD) and dry (-35 cm WTD) cores after 9 months of dry treatment in the laboratory trial (**top**) and the corresponding calculated changes in soil organic carbon (SOC) stocks (**bottom**) over 15 cm depth for the different vegetation groups, and for the three vegetation groups combined (mixed). These are compared with the cumulative loss of SOC calculated for different lengths of time based on the reported 2% annual carbon loss estimates by Bellamy et al. (2005) for organic soils (see main text for details). Significant differences (ANOVA) are indicated by different letters.

Notwithstanding the lack of a consistent difference in BD between WTDs, the changes in BD under the three vegetation types were used to predict potential impacts on C stock estimates (**Figure 98**; bottom). Assuming that BD is constant (using an average of 0.11 g cm^{-3} as in this study) over the 15 cm depth sampled by Bellamy et al. (2005), the observed BD differences (**Figure 98**; top) corresponded to an average difference of 10.8% in the amount of soil organic carbon (SOC) calculated for wet and dry cores, with the difference for *Calluna* being 12.5%, for *Sphagnum* 30.3% and for *Eriophorum* -23.7%. Taking the average original C content in the Bellamy et al. (2005) study for the >300 g/kg soils group (i.e. organic soils), which was 439.7 g kg^{-1}, and (assuming constant BD between sampling of 0.11 g cm^{-3}) subtracting their reported 2% change per year cumulatively over 12 years (the minimum gap between samples in their study), soils would have lost 21.5% of their SOC, a value equivalent to 1.39 kg C m^{-2} over a depth of 15 cm. Over a period of 25 years (maximum sampling gap in their study), soils would have lost 39.7% of their C, a value equivalent to 2.62 kg C m^{-2} over 15 cm. The change based on the significant difference between wet and dry BD for the *Sphagnum* dominated peat was 1.99 kg C m^{-2} over 15 cm (**Figure 98**; bottom), which is between the two calculated values for the minimum and maximum C loss for the data reported in Bellamy et al. (2005).

Therefore, based on the drought-induced changes in BD from the laboratory cores, the reported apparent change in soil C stocks in Bellamy et al. (2005) is well within the calculated possible changes due to natural BD changes. The BD measurements in this study were taken from a relatively narrow range (albeit over an extended period) of WTD of 0 cm and 35 cm, whilst the range of WTDs in the sampling programme of Bellamy et al. (2005) might have been even lower, reflecting not only changes in climatic conditions but also management. A feasibility assessment of the potential magnitude of a peat expansion effect is possible by calculating the change in peat depth implied by the findings of Bellamy et al. (2005), assuming there were no changes in BD. Taking the original Bellamy et al. (2005) C content (439.7 g/kg soil) and dividing it by 15 cm to give 29.3 g kg^{-1} soil cm^{-1}, and using their value of 7.37 g kg^{-1} soil yr^{-1} loss in soils of a high C content (cf. Table 1 in Bellamy et al., 2005), this corresponds to a "loss" of ~0.25 cm yr^{-1}. This "loss" equates to ~3 cm over 12 years and 6.3 cm over 25 years. The former is well within the changes in peat depth observed here (**Figure 97** and also **Figure 101** below) purely by natural fluctuations in peat shrinkage and expansion (most likely in relation to WTD changes), the latter also if changes in BD are accounted for (**Figure 98**; bottom).

Field assessment (see **Figure 95**; right):

In August 2014, steel rods of 1.2 cm diameter were hammered into the bedrock/clay layer to assess changes in the peat surface level. At all three sites, one rod was inserted next to the WTD logger in each 5 x 5 m monitoring plot. This gave 24 rods at each site, split between the main four management treatments (i.e. combined *Sphagnum* addition for mown plots). Additionally, at Mossdale a further 90 rods were installed in sets of three within five different vegetation/management types around a manual WTD dipwell hole. The five areas were dominated by *Sphagnum*, *Calluna* on unmanaged, burnt or mown areas and *Eriophorum*. For each group, three sets of three rods were on flat slopes (≤ 5°) and another three sets on steeper slopes (> 5°). The offset from the top of the rod to the peat surface, as well as the WTD at each location, was measured on six occasions during 2014-2015 across all seasons. A groove 20 cm from the top of each pole was cut prior to installation and the distance from the top of the pole to the groove was measured to check for expansion or shrinkage of the metal rod (in response to temperature changes), but none was detected. Bulk density was determined for three depths (0-5 cm, 10-15 cm and 25-30 cm) in August 2014 for each core removed to create the hole for the dipwell.

For the plot data, the greatest range of vertical movement in the peat height on a single plot was 1.9 cm at Nidderdale, 2.3 cm at Mossdale and 2.7 cm at Whitendale, with the smallest range being 0.4 cm at all sites. Overall, the average vertical movement in peat depth for the plot data was 1.1 cm with a median of just under 1 cm (**Figure 99**), which corresponded well to the observed values in the laboratory experiment (**Figure 97**). There was a significant effect of change in WTD on the change in peat depth ($F_{1, 410}$ = 23.36, p< 0.0001), with an increase in peat height being related to a higher WTD (i.e. wetter). There was also a significant effect of management on change in peat height ($F_{3, 410}$ = 4.76, p = 0.0028) and a significant interaction between site and management ($F_{6, 410}$ = 6.82, p< 0.0001). Only at Mossdale was the mean change in peat depth on burnt plots significantly more negative (i.e. either greater shrinkage or less expansion; p< 0.0001) than under other managements (**Figure 99**).

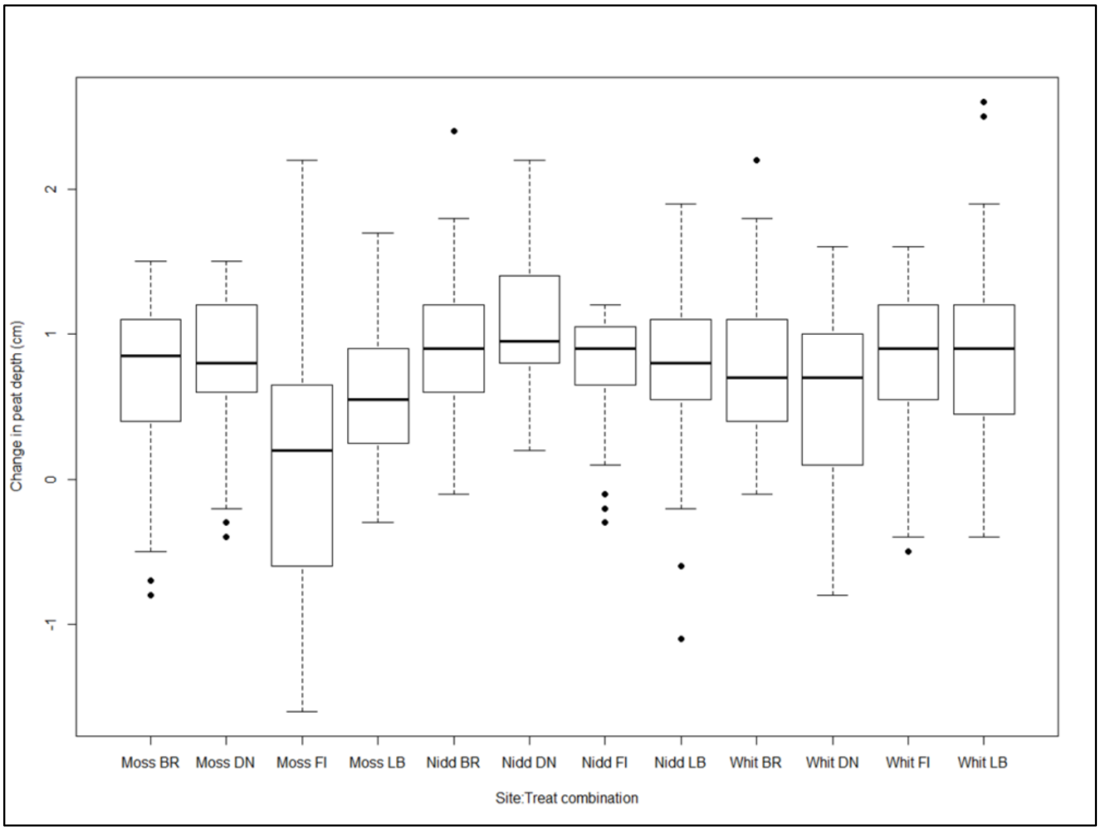


Fig. 99 Shrinkage and expansion ranges of peat depth (in cm compared to initial peat levels) measured in the field under naturally fluctuating water tables for different managements (n = 4) at Mossdale (Moss), Nidderdale (Nidd) and Whitendale (Whit) between August 2014 and December 2015. Management codes were DN (uncut), BR (brash removed), LB (left brash) and FI (burnt). Thick lines indicate medians, boxes show the interquartile ranges and outliers are shown as dots.

For the additional rods within different vegetation and management at Mossdale the average range of the change in peat depth was 1.3 cm, the smallest range was 0.2 cm and the largest 3.1 cm across the different vegetation patches and slopes, with the median depth change being close to 0 cm (**Figure 100**). There was a small significant effect of vegetation/management type on the change in peat height ($F_{4, 167} = 2.97$, $p = 0.0210$), although only peat height change between *Calluna*-covered ground and burnt ground was significantly different ($p < 0.02$).

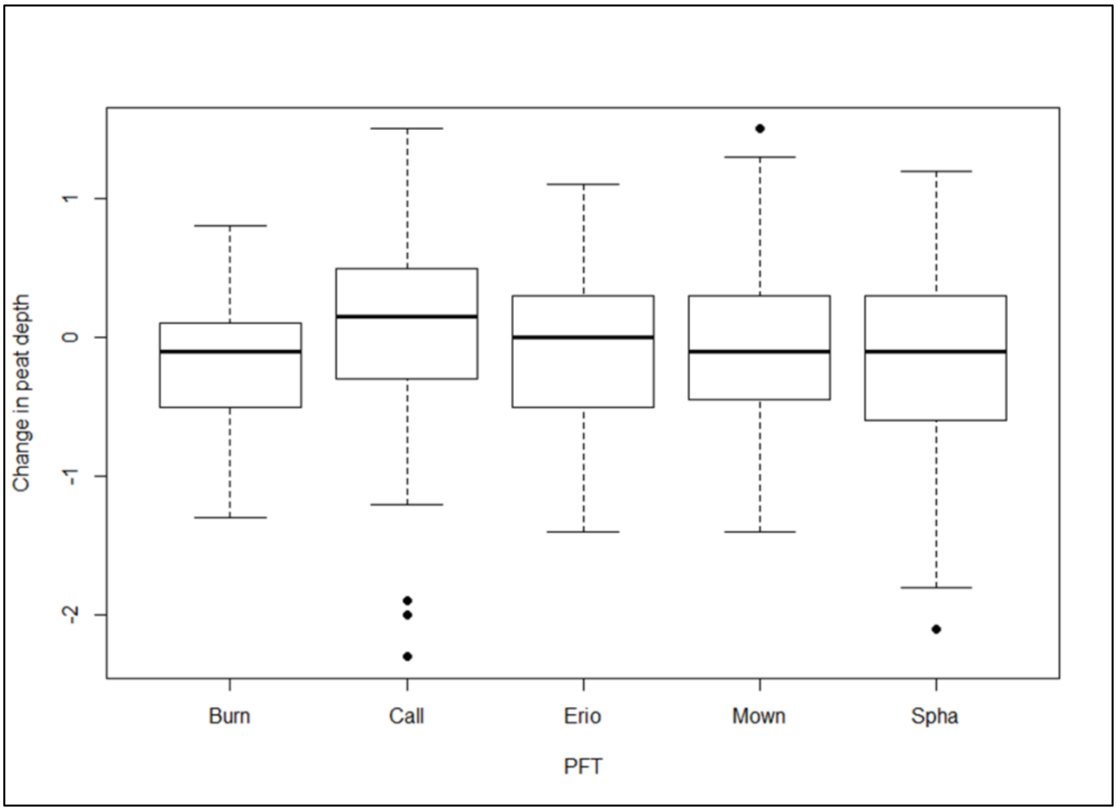


Fig. 100 Shrinkage and expansion ranges of peat depth (in cm compared to initial peat levels) measured in the field under naturally fluctuating water tables for the additional plots at Mossdale (n = 18 each) for burnt *Calluna* (Burn), mown *Calluna* (Mown), unmanaged *Calluna* (Call), *Eriophorum* (Erio) and *Sphagnum* (Spha) dominated areas between August 2014 and December 2015. Thick lines indicate medians, boxes show the interquartile ranges and outliers are shown as dots.

There was a significant positive effect of change in WTD on the change in peat height for the extra rods at Mossdale ($F_{1, 168} = 6.44$, $p = 0.0121$) (**Figure 101**). The slope also significantly affected the change in peat height ($F_{1, 168} = 5.55$, $p = 0.0197$), with increasing slope (x) causing increased depth changes (y) as evident in the fitted exponential relationship across all Mossdale peat depth data of $y = 122.55*e^{-0.045x}$ ($R^2 = 0.43$).

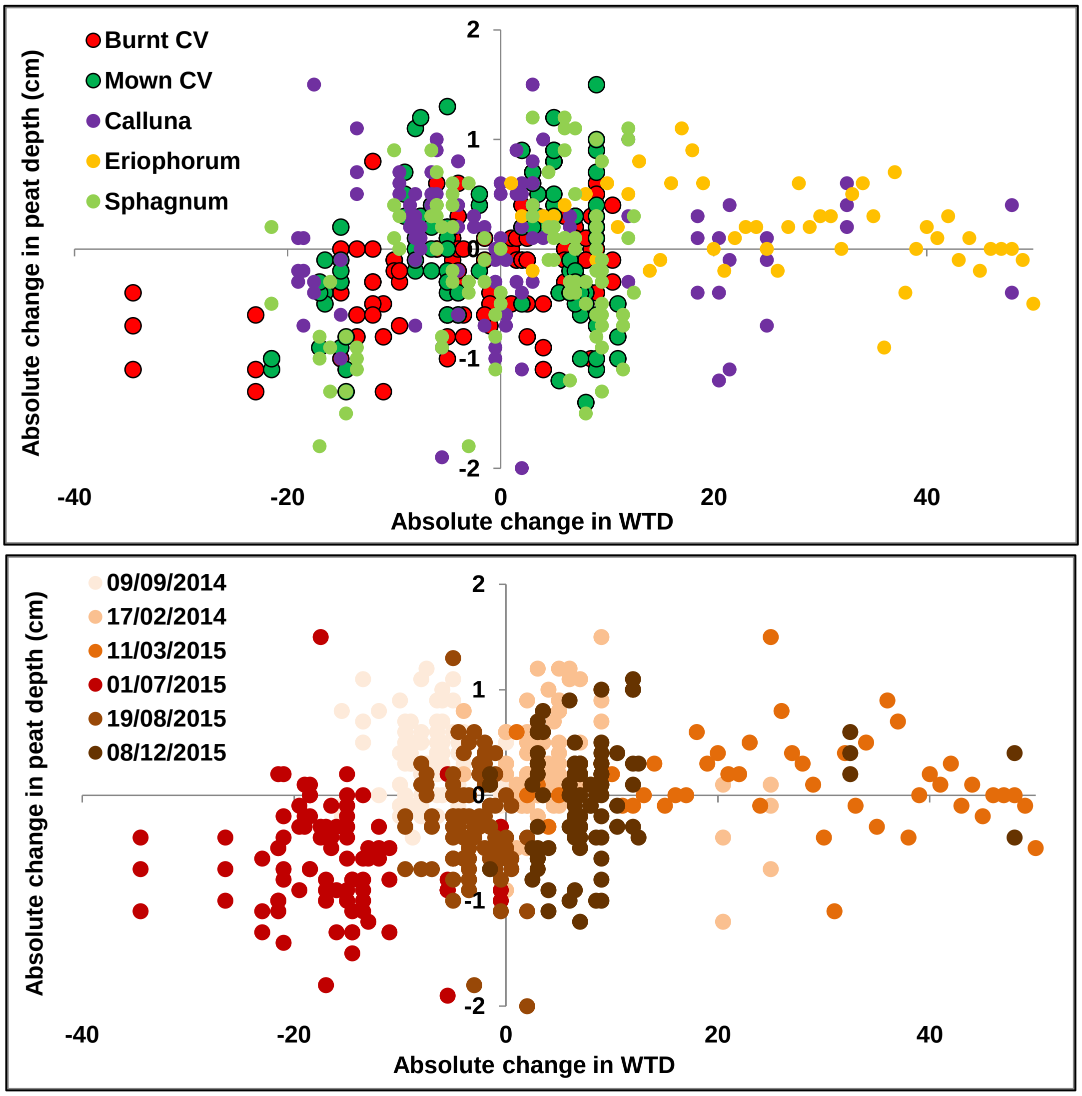


Fig. 101 Shrinkage and expansion ranges as the absolute change (in cm) compared to initial peat levels as measured in the field between August 2014 and December 2015 under naturally fluctuating various water table depth (WTD) for the additional plots at Mossdale **(top)** for the main management and vegetation areas of burnt *Calluna* (Burnt CV), mown *Calluna* (Mown CV), unmanaged *Calluna*, *Eriophorum* and *Sphagnum* dominated, and **(bottom)** for the individual monitoring dates.

Finally, the installed peat rods offer an exceptional long-term monitoring platform for actual peat growth across three sites in Northern England. This simple tool will allow monitoring by a lay person, such as a local farmer or gamekeeper, to enable attainment of practitioner-relevant information on the state of the peatland (i.e. net peat growth rates over time). Information on peat depth and changes over time is an essential component of carbon accumulation calculations (Gorham, 1991) and is also used in developing carbon inventories (Parry & Charman, 2013). If these sites were to be maintained beyond one management rotation, this would allow the first assessment of a long-term landscape scale impact of different grouse moor management, as well as climate impacts, on actual peat accumulation rates in the UK in relation to a fixed datum. Such data will be invaluable for any development, as done by Smyth et al. (2015), or the validation of future carbon offsetting schemes such as the IUCN UK's Peatland Code.

In summary, the laboratory and field comparisons reported here supported the two initial hypotheses, firstly, that 'apparent' changes in C_{org} densities can be explained by peat surface (depth) fluctuations, therefore highlighting the need to consider BD changes when reporting (surface) C stocks as was not done in Bellamy et al. (2005); secondly, that the peat surface and bulk density fluctuations can be affected by land management, vegetation type and WTD changes. Moreover, the differences in recovery rates after shrinkage due to drought between vegetation types highlighted the crucial role *Sphagnum* has in creating a more drought resilient peat matrix.

4.4.2 *Decomposition rates*

The field assessment of soil respiration (SR) from decomposition (cut areas) did not reveal a significant management effect on fluxes (**Figure 72**) or on their temperature sensitivity (**Table 14**), although it indicated reduced SR from burnt plots. However, field measurements included the entire peat column, which might limit the potential of flux measurements in the field to detect the effects of a change in recent management largely affecting surface layers. Moreover, whilst field decomposition fluxes showed a strong temperature response (Section 4.2.13), the range of soil moisture values during field measurements was too small to robustly derive any relationships. Again, surface peat layer responses might differ to that of lower peat layers, therefore hiding management impacts on environmental drivers of SR in relation to recent management change. Notably, previous peat incubation experiments (Wang et al., 2010) show a strong non-linear (polynomial) relationship between moisture and temperature. Therefore, a further investigation based on more shallow peat samples was conducted in a controlled laboratory study (**Figure 102**), specifically comparing temperature and moisture impacts on decomposition fluxes from burnt and mown peat samples over a period of five months. More detail of these laboratory studies is provided in Appendix 7. The experiment addressed the following hypotheses:

1) SR will be lower from burnt than mown samples, but only in surface layers, whilst deeper layers do not show a significant difference.
2) Moisture and temperature responses will be different between peat layers and differ between burnt and mown samples only for surface layers with no differences between deeper peat layers.

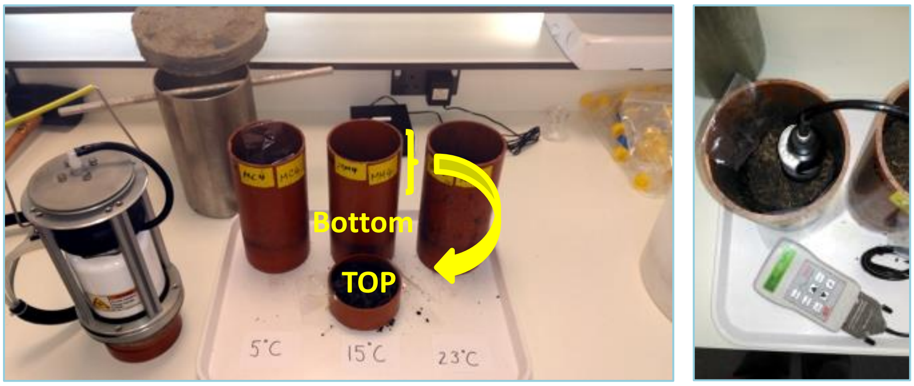


Fig. 102 Peat samples were collected in 25 cm deep PVC tubes (**left**) from burnt and mown sites at all three sites (Nidderdale, Mossdale and Whitendale) and incubated at three different temperatures (4, 15 and 23 °C) with soil moisture (monitored with a SM200 Delta T probe; **right**) declining over time. Fluxes were monitored using a Li-Cor 8100 soil chamber (10 cm diameter) for the entire core and separately for the top 5 cm layer (the difference between the full core and the top was equal to the flux from the bottom section).

Peat samples were collected on 1st April 2016 using 25 cm deep (10 cm diameter) PVC tubes, which were suitable for monitoring SR with the LiCor soil flux chamber (**Figure 102**). At all three sites, four cores were taken each from burnt and mown with brash left (LB) plots. Any surface vegetation was cleaned from the cores (i.e. green moss and any shoots were removed) before they were sectioned into a 5 cm 'top' layer and the 'bottom' 15 cm of peat. A perforated custom-made plastic sleeve allowed the top section to be lifted and replaced for separate flux measurements (**Figure 102**). The cores were incubated in open buckets with a stable water table in the dark under different temperatures so that soil moisture could change naturally over time. The temperature was initially 4°C and the water table 0 cm (i.e. dry) for all samples. One replicate per site from each management was then allocated to one of three temperatures (4°C 'cold'; 15°C 'mid'; 23°C 'warm'). The water table was kept at 6 cm (from the bottom) throughout but was lowered to 0 cm for the last two months. Allocation of replicates to a temperature treatment was based on ranking the mean flux over three measurements during the pre-period of cold conditions (4°C). However, one sample for each management was kept at the initial conditions. Additional chemical analyses were performed, as described in the following Section 4.4.3. Moisture was measured using a SM200 probe (Delta T Devices, UK).

The mean SR across the three sites showed significantly ($p<0.001$) lower SR for bottom than top core sections and also significantly ($p< 0.05$) lower SR on burnt plots but only for the 23°C treatment (see **Figure A7.1** in Appendix 7; although also lower on the burnt, the difference for the 4°C treatment was marginally significant at $p = 0.082$).

Overall, the three sites revealed very similar temperature and moisture responses. Whereas fluxes increased exponentially with increasing temperatures (**Figure 103**; left), there was a polynomial relationship between soil moisture and fluxes, with fluxes decreasing below about 60% soil moisture (**Figure 103**; right). Moreover, both temperature and moisture effects differed between the peat sections. Decomposition fluxes from both managements increased less steeply as temperature increased for the surface 5 cm sections.

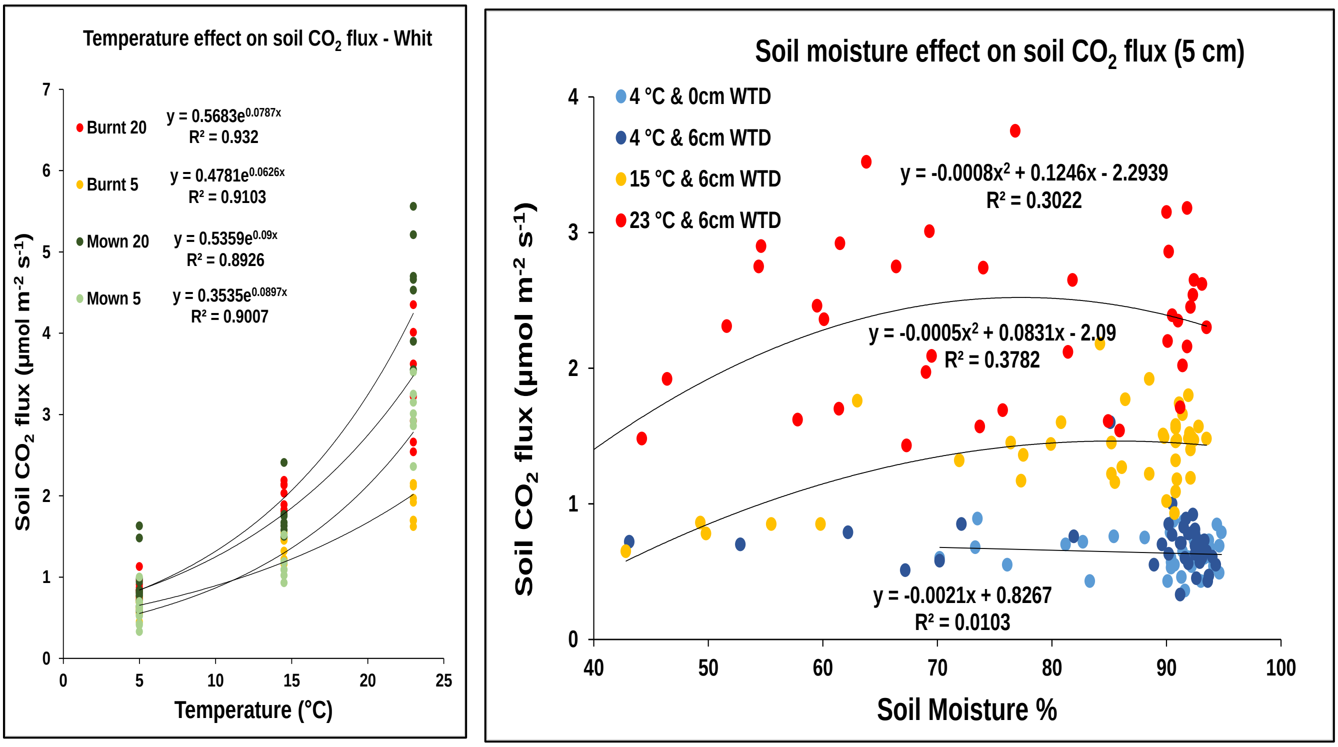


Fig. 103 Peat core section (top 5 cm section or 20 cm total core) decomposition fluxes from incubations at different temperature (**left**; from Whitendale) and moisture (**right**; from all sites for the top 5 cm section only). Temperature incubations were at 4, 15 or 23 °C and water table depths either at 0 cm or 6 cm. Regression lines and equations are shown.

Consequently, calculated temperature sensitivity (Q_{10}; see Section 4.2.13 and Appendix 6) was generally higher for the bottom section and the average Q_{10} increased from the top (2.1) to the total core (2.4) to the bottom (3.1) sections (**Table 22**). These Q_{10} values for total cores were only slightly lower than the field estimates of Q_{10} (~3.2) for SR decomposition fluxes when roots were cut (**Table 14**). There was no consistent difference between sites, although Nidderdale samples tended to show the lowest Q_{10} values (**Table 22**). Whereas there was no clear management effect on Q_{10} for the bottom sections (which had very large SEs), the total cores and particularly the top sections (with the lowest SE) had a lower Q_{10} for the burnt cores, especially when considering Nidderdale and Whitendale separately (**Table 22**); this effect was likely to be significant, based on the SEs not overlapping (see Heinemeyer et al., 2012). These are the first data on temperature sensitivity for separate peat layers and reveal a management impact on SR and Q_{10} values. Importantly, whilst cores from mown areas where the brash was left showed higher temperature sensitivity, likely reflecting the availability of substrate, burnt surface cores had lower Q_{10} values (below 2) in addition to observed lower overall SR fluxes in the field (**Table 14**). This reduction in temperature sensitivity of decomposition in peat from burnt plots could relate to either charcoal bypassing decomposition of soil carbon inputs, as proposed by Clay at et al. (2010b), and/or supressing decomposition of soil organic matter by altering soil chemical, physical, and biological properties and processes (Pingree and DeLuca, 2017).

The reduction in soil moisture was related to the peat naturally drying out, particularly under warmer temperatures, over five month incubation, from August 2015 until December 2015. This lowering of moisture is a natural process in which the peat (surface) dries out during summer months when there is no significant rainfall. Indeed, the lowest soil moistures recorded in the field were around 30%, similar to those in the incubation study for the top layer (**Figure 103**). Therefore, together with similar temperature ranges, the incubation achieved a realistic range of environmental conditions. However, no clear site-specific soil moisture responses were observed. Soil moisture did affect the contribution of the top 5 cm to decomposition fluxes, which were very high initially with 60% for Mossdale and 75% for both Nidderdale and Whitendale (data not shown). However, this decreased over time with increasing temperature, but only once moisture was less than 60%, to around 30-50%, contribution of the top 5 cm layer.

Table 22 Temperature sensitivity (Q_{10} values ±95% confidence intervals, SE, df (all P-values were <0.001)) for peat core decomposition rates, either for the top (5 cm) section, the entire core (20 cm), or the bottom (15 cm) section only for either the individual sites, all sites or Nidderdale (Nidd) and Whitendale (Whit) combined without Mossdale (Moss) for the two managements for peat samples taken from the burnt (B) versus mown (M) sub-catchments (taken three years after the initial management).

Individual sites

Q10 table	Q10	±95%	±SE	DF
Moss_B_5cm	2.19	0.28	0.14	26
Nidd_B_5cm	1.82	0.25	0.13	26
Whit_B_5cm	1.87	0.15	0.08	26
Moss_M_5cm	2.10	0.16	0.08	26
Nidd_M_5cm	2.17	0.17	0.09	26
Whit_M_5cm	2.45	0.30	0.15	26
Moss_B_20cm	2.46	0.28	0.14	26
Nidd_B_20cm	2.20	0.30	0.15	26
Whit_B_20cm	2.20	0.19	0.10	26
Moss_M_20cm	2.46	0.20	0.10	26
Nidd_M_20cm	2.35	0.18	0.09	26
Whit_M_20cm	2.46	0.32	0.16	26
Moss B Bottom	3.17	0.83	0.42	26
Nidd B Bottom	2.98	0.85	0.43	26
Whit B Bottom	3.23	1.01	0.52	26
Moss M Bottom	3.18	0.71	0.36	26
Nidd M Bottom	3.01	0.79	0.40	26
Whit M Bottom	2.66	0.73	0.37	26

All sites

Q10 table	Q10	±95%	±SE	DF
B_5cm	1.95	0.13	0.07	80
M_5cm	2.24	0.13	0.06	80
B_20cm	2.28	0.16	0.08	80
M_20cm	2.42	0.16	0.08	80
B_Bottom	3.13	0.53	0.27	80
M_Bottom	2.94	0.47	0.24	80

Nidderdale & Whitendale only

Q10 table	Q10	±95%	±SE	DF
B_5cm	1.84	0.14	0.07	53
M_5cm	2.31	0.17	0.09	53
B_20cm	2.20	0.17	0.09	53
M_20cm	2.41	0.17	0.09	53
B_Bottom	3.10	0.65	0.33	53
M_Bottom	2.83	0.51	0.26	53

The obtained data on responses of decomposition flux to both temperature and soil moisture (SM) were used to fit a polynomial model, following Wang et al. (2010). When averaged across all sites, fluxes increased with increasing temperatures for all peat core sections but there were quite different relationships with SM (**Figure 104**). Notably, whereas the top layer showed a clear reduction in SR fluxes at lower values of SM, the bottom core showed little relationship between fluxes and SM; the SM ranges were much less in the bottom core. However, when looking at individual sites (see polynomial equations in **Table A7.1** and **Figure A7.4** in Appendix 7), the whole core often showed a slightly different SM relationship when based on SM values measured in the peat surface, as would commonly be done in the field, as compared to when SM from the top and bottom sections was averaged (weighted by depth).

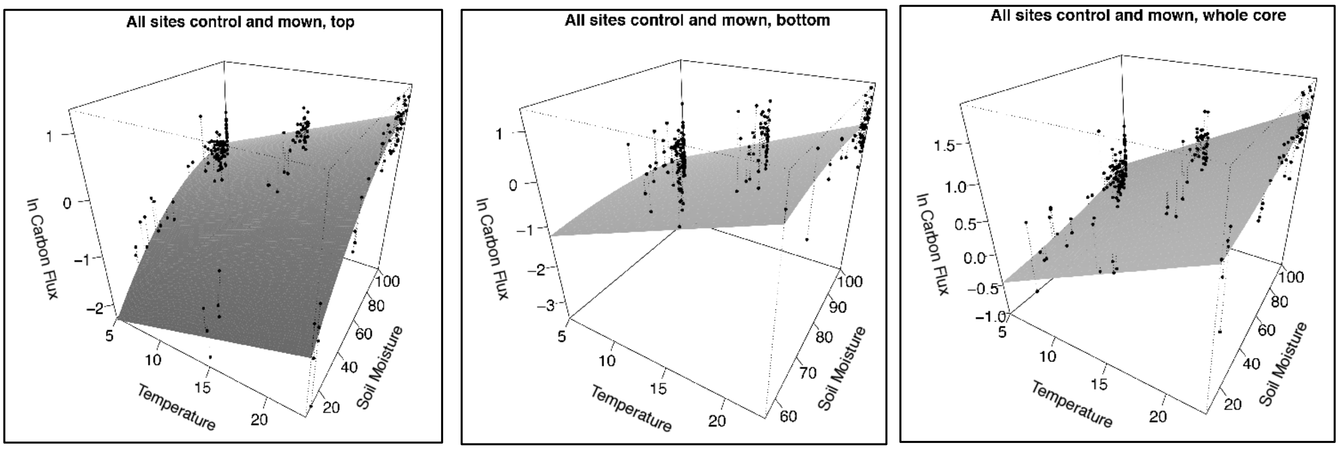


Fig. 104 Peat decomposition (flux) data and their distances (dotted lines) to the fitted polynomial surfaces (see **Table A7.1** in Appendix 7 for the parameters and statistics) for the averages of all sites for the top 5 cm section (**left**), the bottom 15 cm section (**middle**) and total 20 cm core (**right**). The natural log (ln) of the soil respiration fluxes (z-axis) were plotted against temperature (x-axis) and percentage soil moisture (y-axis) for the combined data for both managements of burnt (control) and mown. Note the different y-axis and z-axis scales.

Comparing managements across all sites showed a much stronger SM limitation (over a similar SM range) on mown compared to burnt cores (**Figure 105**), which was largely related to changes in the relationship in the top layer (see **Figure A7.4** in Appendix 7 for a full set of graphical analysis output).

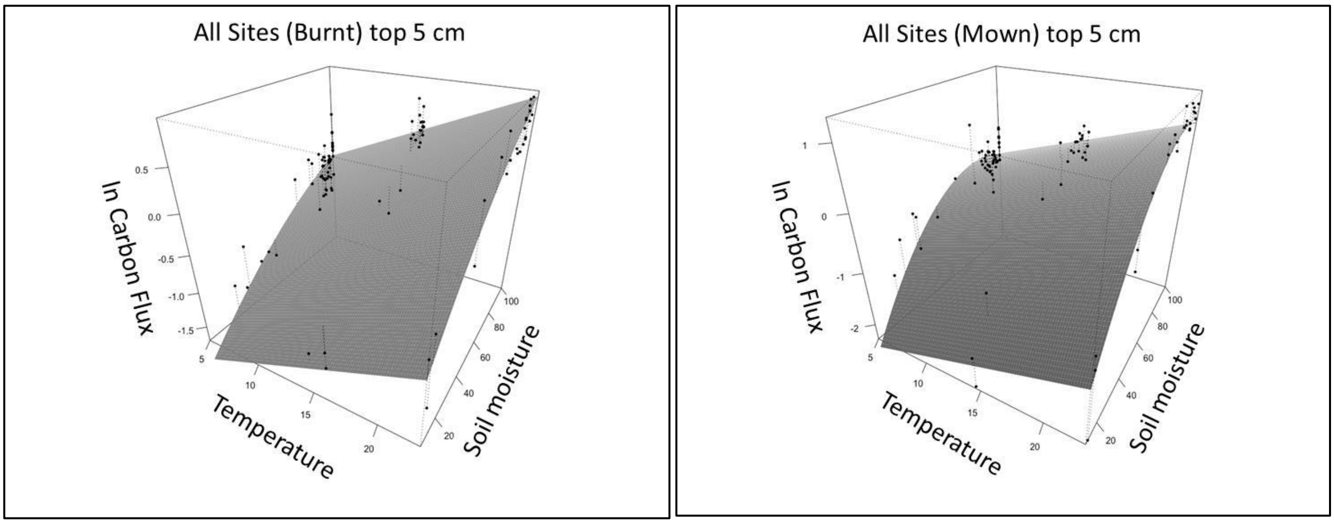


Fig. 105 Peat decomposition (flux) data and their distances (dotted lines) to the fitted polynomial surfaces (see **Table A7.1** in Appendix 7 for the parameters and statistics) for the averages of all sites for the top 5 cm section of the burnt (**left**) and the mown (**right**) cores. The natural log (ln) of the soil respiration fluxes (z-axis) were plotted against temperature (x-axis) and percentage soil moisture (y-axis). Note the different z-axis scales.

A comparison between the individual sites for combined managements revealed a much stronger top layer response of SR fluxes to SM at both Whitendale and Nidderdale compared to Mossdale, but quite similar bottom and whole core relationships at all sites (see **Table A7.1 and Figure A7.4** in Appendix 7 for a full set of analysis output). This site difference in top sections was mainly driven by stronger SR fluxes responses to changes in SM on mown samples, which was strongest at Nidderdale (see **Figure 106**).

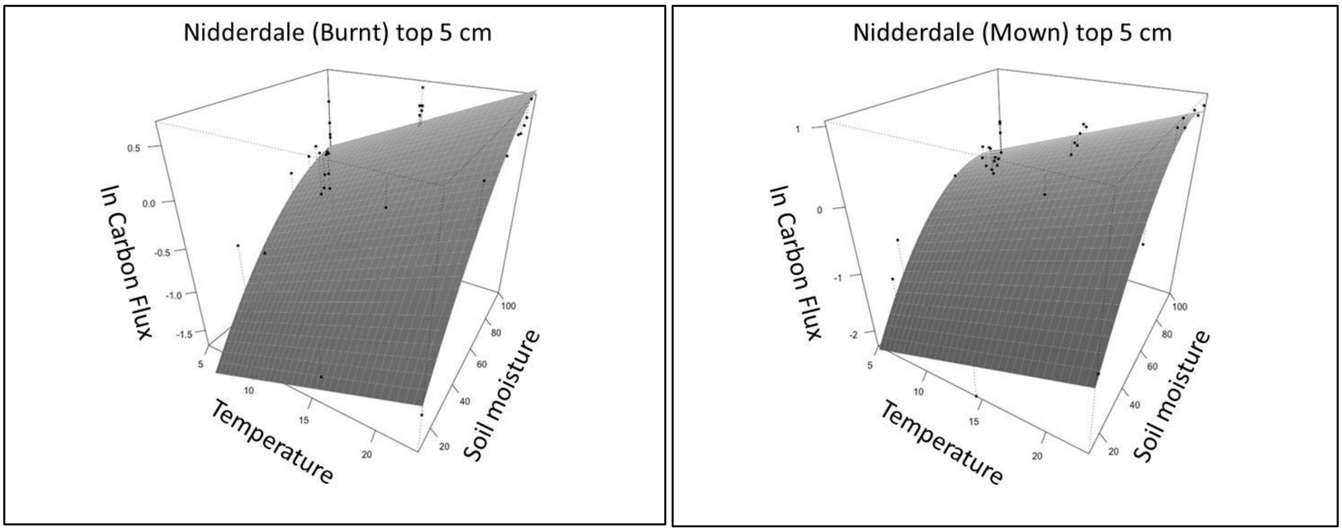


Fig. 106 Peat decomposition (flux) data and their distances (dotted lines) to the fitted polynomial surfaces (see **Table A7.1** in Appendix 7 for the parameters and statistics) for the Nidderdale site for the top 5 cm section of the burnt (**left**) and the mown (**right**) cores. The natural log (ln) of the soil respiration fluxes (z-axis) were plotted against temperature (x-axis) and percentage soil moisture (y-axis). Note the different z-axis scales.

Comparison to published data by Wang et al. (2010) for a permafrost peatland site in China showed a surprisingly good agreement in the moisture and temperature coefficient parameter estimates (see **Table A7.1** in Appendix 7). Overall, the intercept was -2.4 (compared to 2.4 by Wang et al., 2010) with a temperature coefficient of around 0.08 (cf. 0.11) and a SM coefficient of 0.05 (cf. 0.02) and a SM^2 coefficient of -0.0001 (cf. -0.0001). The higher fluxes (i.e. positive intercept) reported by Wang et al. (2010) likely reflect their sieved and homogenised samples. To obtain such similar response parameters for temperature and moisture effects for very different peat samples was surprising. However, peat decomposition in the Chinese ombrotrophic bog might well be similar to the UK blanket bogs, considering the similar shrub and moss vegetation (Wang et al., 2010).

In summary, the most striking findings from this laboratory experiment was the significantly higher overall surface SR flux contribution of around 70% under higher than 60% SM and its strong response to SM changes, which was significantly higher on mown than on burnt plots overall (supporting hypothesis 1) and lowest at the wettest site, Mossdale. Temperature sensitivity (Q_{10}) values did not differ between sites but were lower for burnt than mown surface peat, without any Q_{10} differences in deeper layers (supporting hypothesis 2). However, we acknowledge that in addition to, or instead of, the hypothesised charcoal effect, decomposition processes could also have been reduced in the burnt cores because of the loss of biomass (via combustion) and reduced plant-driven C cycle processes and thus the supply of labile organic matter to the soil microbes.

4.4.3 *Peat chemistry*

The differences in decomposition rates observed in section 4.4.2 were investigated further by chemical analysis taken from the same peat samples (burnt versus mown peat samples from the three sites). Rather than an in depth analysis using one method, a multi analysis approach (see **Figure 107**) was chosen to reveal the most promising approach for future analysis. Appendix 7 provides a full description of the analyses and further results.

Fig. 107 Example of sampling for chemical peat properties from the incubated peat samples (see Section 4.4.2). Three ~1 cm^3 samples (one from the top, one at 6 cm depth and one from the bottom) were taken (two pictures on the left) for analysis by various analytical methods: (from left to right) digestibility assay (using destructive UV spectrophotometry in the Centre for Novel Agricultural Products), ^{13}C NMR, FTIR and Py-GC/MS.

Peat chemistry is not well understood, although peat models base predicted carbon stock changes and greenhouse gas emissions on existing published data on peat chemistry classes (e.g. Heinemeyer et al, 2010). Lignin is assumed to be the most resistant component of plant residues entering the soil (Killham, 1994). In almost all studies, the contribution of lignin to peat SOM increases with depth, identifying it as a recalcitrant component (Cocozza et al., 2003), although lignin still decomposes down the peat profile (Mason et al., 2012) producing esterified, depolymerised phenolic derivatives (Killham, 1994). By contrast, carbohydrate content decreases with depth (Jörgensen & Richter, 1992), with polysaccharides degrading more quickly than cellulosic carbohydrates (Kaal et al., 2007). Trends for lipids are less clear, possibly as a result of their much lower concentrations and more complex nature. Hoyos-Santillan et al. (2015) found that long chain fatty acid concentrations decreased with depth, but no clear pattern was observed for short chain fatty acids. Increasing lipid concentrations with depth (Kaal et al., 2007) and selective preservation of lipids under anoxic conditions (Zaccone, 2007) have also been observed. Pyrogenic (black) carbon, which exists as a continuum from charred biomass to highly condensed graphite-like materials, is considered to be highly resistant to degradation (Schmidt & Noak, 2000). Therefore, the production of black carbon by managed burns might increase the recalcitrance of peat organic matter. No study has specifically considered the chemical effects on peat of managed burns of heather moorland, although one study was found on changes in pH and a range of nutrients over the course of a 20 year burn cycle (Rosenburgh et al., 2013). However, one PhD study found burnt tropical peatlands had higher labile carbon constituents than unburned peats, with significant amounts of charcoal in the upper 5 cm of the burned peat (Milner, 2013).

The knowledge of peat chemistry summarised above is based on old methods of chemical analysis. Novel analytical methods, such as digestibility assays in combination with destructive UV spectrophotometry, solid state Nuclear Magnetic Resonance spectroscopy (^{13}C NMR), Fourier-transform infra-red spectroscopy (FTIR) and pyrolysis gas chromatography mass spectrometry (Py-GC/MS), offer exciting opportunities to address this knowledge gap. Most differences in microbial activity - and thus changes in peat chemical composition - can be expected at the surface peat layer, where microbes will be most active under an intermediate water table and soil moisture content. Microbial activity will be enhanced further by warmer temperatures and an available substrate supply (e.g. litter inputs). Microbial activity slows under very dry conditions, creating a bell-shaped curve of decomposition rates versus soil moisture under warm conditions. In contrast, cold conditions should buffer these

changes and lead to a more linear decomposition response as soil moisture also remains fairly constant, as demonstrated by the results in Section 4.4.2.

Any differences in decomposition rates should cause changes in composition and concentrations of key peat chemical groups. However, there are currently no published data available on the impacts of soil moisture and temperature on the chemical nature of peat, particularly at different depths. Such data would aid validation of model assumptions and enhance understanding of decomposition processes. A particular issue with soil carbon models is the arbitrary representation of the many carbon compounds found in soils in only a few carbon pools. For example, the MILLENNIA peat development model (Heinemeyer et al., 2010) considers the "cellulose" fraction to contain mostly carbohydrates, whilst the "soluble" fraction captures all other carbon except that in the "recalcitrant" lignin or cellulose pools and thus is assumed to be associated with easily degradable aliphatic and protein compounds. However, both data of peat chemical composition and validations of model assumptions of carbon pools are rare and pose a considerable knowledge gap in process-level understanding towards improved model representation of soil carbon cycling. The peat chemical analyses performed on the peat samples form the three sites tried to provide data on the proportional abundance of key chemical groups in peat carbon in relation to depth, which could be compared to the modelled carbon pools. Moreover, the analysis tried to capture site and management differences in light of decomposability. The overall hypothesis was that recent management change affected the chemical composition of the peat at the surface, whilst the previous burn management resulted in a fairly similar chemistry further down the peat profile.

Results

The digestibility assay analysis, together with destructive UV spectrophotometry (analysed at the Centre for Novel Agricultural Products; York), provided estimates for the three sites of crystalline cellulose and lignin contents for each peat depth (0, 5 and 20 cm) in relation to total carbon content, together with soluble carbon and ash contents (**Table 23**). All sites had a similar crystalline cellulose content of around 15% and a lignin content of around 25% (**Figure 108**). Whereas cellulose decreased with depth to around 10%, lignin increased to around 30%. There was no significant difference at any depths between samples from burnt or mown plots. However, when data from all depths were combined, increasing temperature caused a significant ($p < 0.01$) increase in lignin content as a percentage of total carbon content (mean of 23% at 4°C versus 27% at 23°C), which is consistent with lignin being a relatively recalcitrant carbon compound. Ash content was around 2% whilst the soluble fraction was less than 1%. However, as a considerable amount of carbon is lost in the extraction process using the digestibility method, a correction was applied to allocate this 'lost' carbon equally to soluble and cellulose fractions (**Table 23**). This equal allocation reflected that the extraction method related loss would have not discriminated between the soluble and cellulose fractions (i.e. equally prone to be flushed out). This increased the soluble fraction to around 30% and the cellulose component to 40%.

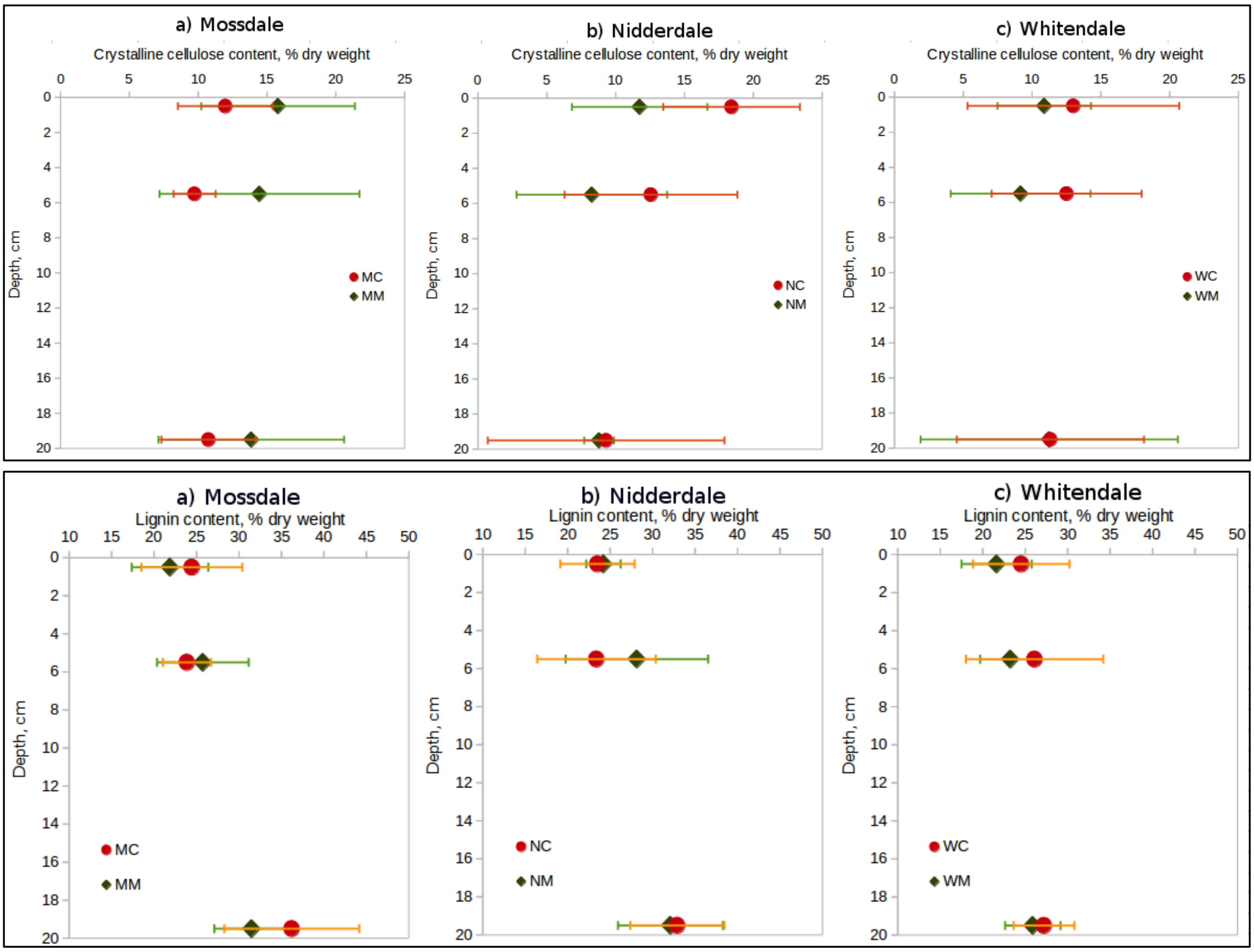


Fig. 108 Mean (± standard deviation) of the percentage of crystalline cellulose (**top**) and the percentage of lignin (**bottom**) (polysaccharides were <1%) obtained by digestibility analyses (using destructive UV spectrophotometry in the Centre for Novel Agricultural Products) for peat samples at 1 cm, 5 cm and 20 cm depth. Orange circles are for burnt (C) samples and green circles are for mown (M) samples, with samples taken from the three sites (Nidderdale (N), Mossdale (M), Whitendale (W)). Peat samples taken on 01/04/16 were analysed on 18/10/16 after 8 weeks of incubation at different temperatures.

Table 23 MILLENNIA model predictions of the main carbon fractions (soluble, cellulose and lignin) for three layers of the top 20 cm of peat (**left**) compared with chemical fractions aggregated into various chemical groups, measured either using destructive UV spectrophotometry in the Centre for Novel Agricultural Products (middle part of table; values in brackets are adjusted to account for the 'lost' carbon fractions, i.e. to add up to 100%, which were added equally to the Soluble and Other and Cellulose fractions) or measured using ^{13}C solid state NMR spectra estimates (right part of table; only measured for the top peat layer). Note the CNAP analysis was done for samples from all three sites, Nidderdale (Nidd), Mossdale (Moss) and Whitendale (Whit), whilst the NMR analysis was performed only for Mossdale (Moss) samples.

Depth	Moddled C fraction (%)			Measured C fraction (%) CNAP Nidd-Moss-Whit				Measured C fraction (%) NMR for Burnt/Mown (Moss)					
(cm)	Soluble	Cellulose	Lignin	Soluble & Other	Cellulose	Lignin	Ash	Carbohydrate	Protein	Lignin	Alliphatic	Carbonyl	Charcoal
1.5	23	42	35	<1 (31)	15-15-15 (45)	20-22-25	2	42/40	9/8	24/31	17/18	5/4	3/0
10	24	33	42	<1 (31)	12-12-12 (42)	25-26-24	2						
20	25	32	43	<1 (28)	12-12-12 (40)	29-34-26	2						

The C fractions were estimated by ^{13}C solid state NMR for peat surface samples at Mossdale (**Table 23** above). Samples from burnt areas showed significantly ($p < 0.05$) higher lignin concentrations of ~24% compared to 31% on mown plots. Carbohydrate and aliphatic content was also high, but was similar between managements, with

about 40% carbohydrate and 18% aliphatic content. Smaller contributions were made by protein (8%) and carbonyl (5%) components. Notably, only the burnt surface samples contained charcoal (3%).

When used for Mossdale, the MILLENNIA model (Heinemeyer et al., 2010) estimated that the "cellulose" fraction averaged 35%, and decreased with depth, and the "soluble" and lignin fractions averaged 24% and 40%, respectively, and particularly lignin increased with depth (**Table 23**). These results were broadly similar to those measured by the analytical methods, especially when considering the corrected digestibility values for the soluble and cellulose fractions. In particular, a more detailed non-destructive NMR analysis over depth and between sites and management could provide vital insight into the accuracy of the model-assumed carbon fractions. For example, the only significant differences between management were observed using NMR for lignin and charcoal, with a significantly ($p = 0.025$) larger lignin fraction in mown samples but higher ($p = 0.031$) charcoal amounts in burnt samples. This seems to agree with the fact that mowing returned brash to the surface, thus increasing lignin amounts during the initial decomposition, whilst the same material was either returned as ash and charcoal on burnt plots, or was lost as smoke and gas during combustion. Overall, whilst the UV spectrometry analysis provided percentage values similar to that of NMR, the NMR analysis was the more robust analysis, both in respect to total amounts of carbon (i.e. avoiding any losses) and detecting differences between managements.

The FTIR analysis of the peat samples across the depth range for peat cores for Mossdale clearly showed spectra peaks in relation to specific C fractions (see **Figure A7.2** in Appendix 7). There were differences between samples at different temperature incubations, which mostly related to changes in surface layers. Moreover, a principal component analysis (PCA) showed separation, based on the C fractions identified, of the burnt and mown surface (1 and 5 cm) peat samples, whilst all of the 20 cm layer samples lacked any clear separation (see **Figure A7.3** in Appendix 7). This supported the overall hypothesis that recent management change affected the chemical composition of the peat at the surface, whilst the previous burn management in both catchments resulted in a fairly similar chemistry further down the peat profile. However, whilst the FTIR technique showed clear analytical potential, further analyses are required to draw more detailed and ecologically meaningful conclusions.

Although the Py-GC/MS samples were not analysed in time for inclusion in this report (machine failure), the analysis has a considerable potential to explain differences in soil respiration (SR) fluxes and temperature sensitivity (Q_{10}) between mown and burnt plots (**Table 14** and **Table 22**, respectively). The observed difference between mown and burnt plots in the NMR results (**Table 23**) and the separation of the top layers by FTIR (**Figure A7.2** in Appendix 7) relates to overall C fractions and the ratios between different chemical compounds. Py-GC/MS has the potential to unravel the chemical composition (Milner, 2013) behind these general differences.

In summary, the findings supported the hypothesis that recent management change affected the chemical composition of the peat at the surface, whilst the previous burn management resulted in a fairly similar chemistry further down the peat profile. The most important differences identified by these chemical analyses were:

(1) the changes in the main carbon fractions with depth, reflecting model predictions of decreasing cellulose and increasing lignin concentrations, and

(2) the higher lignin fraction in mown and the higher charcoal fraction in burnt surface peat samples.

Whilst lignin is a complex chemical group, charcoal also contains many chemical groups due to the incomplete combustion processes. Specific compounds might either stimulate or suppress decomposition, and, linking soil chemistry to process-level studies has a huge potential in explaining such differences. Of particular interest is a potential effect of charcoal in suppressing decomposition fluxes which was observed, for the first time, for rotational burning of blanket bog (see Section 4.4.2). Similar indication for negative priming has been observed previously in crop biochar amendments, linked to DOC absorption (Lu et al., 2014). However, charcoal impacts are complex, sometimes with initial positive priming, followed by specific negative impacts on the decomposition of more complex substances (Cheng et al., 2017).

4.4.4 *Mycorrhizal priming and charcoal effects on decomposition*

Ericoid fungi can form mycorrhizal symbioses with ericaceous species, such as *Calluna vulgaris*, or exist as 'free-living' fungi in the soil. When in a mycorrhizal association, they help the plant to acquire nutrients from the soil and receive carbon from the plant in return (Read, 1991), which benefits the growth of both plant and fungus. Ericoid fungi have saprotrophic properties and it has been demonstrated that they have the capacity to break down structural and recalcitrant components in plant litter and soil, including cellulose (Varma & Bonfante, 1994), hemicellulose (Burke & Cairney, 1998), polyphenols (Varma & Bonfante, 1994) and lignin (Haselwandter et al., 1990). This suggests that, even when they are not in symbiosis, they help to cycle and release nutrients which can aid plant growth, thereby reducing the susceptibility of the peat to erosion (Evans et al., 2014).

Likewise, burning can accelerate plant regeneration (Mahmood et al., 2003), as the residue left behind has a higher content of available nutrients than the soil or unburnt litter. However, burning can also release waxes or bitumens which partially waterproof the peat surface (Clymo & Gore, 1983), thus increasing surface runoff and dissolved organic carbon (DOC) release. However, the overall effect that burning has on water quality is unclear because, although implicated in increasing DOC concentrations (Clutterbuck & Yallop, 2010), burning does not necessarily cause consistent increases in DOC or water colour (Holden et al., 2012; Harper et al., 2018). Additionally, as burning is mainly used on grouse moors in the UK, this cannot explain observed DOC changes in many peat-covered areas across the northern hemisphere (Skjelkvåle et al., 2005).

Any long-term increase in water colour is concerning for water companies, which must abide by regulations to provide water without colour that is safe for human consumption (e.g. in England and Wales - The Water Supply (Water Quality) Regulations 2016). The main cause of colouration is the humic substances which constitute the more hydrophobic components of DOC. Humic substances largely comprise humic acids, which are dark brown to black, and fulvic acids, which are pale brown to yellow (Thurman, 1985). DOC also contains non-coloured substances, consisting mainly of simple hydrophilic compounds such as carbohydrates, fats, proteins and waxes, which are more easily broken down by microorganisms and so have a shorter residence time than the humic substances (Schnitzer & Khan, 1972; Thurman, 1985). DOC is removed from water in treatment plants by adding coagulant, specifically removing the darker humic acids. However, to remove the fulvic acids and non-coloured substances a chlorination process is used which can result in the formation of likely carcinogenic by-products, such as trihalomethanes and haloacetic acids (Singer, 1999; Clay et al., 2012).

Climate scenarios for the UK predict hotter temperatures and a decrease in summer rainfall (Murphy et al., 2009). Such a change in climate is likely to cause shifts in plant assemblages, which on northern peatlands will result in a decline in mosses and other bryophytes to the benefit of vascular plants (Gallego-Sala & Prentice, 2013). It has been suggested that this will be linked to the release of labile C compounds from vascular plants into the soil (Bragazza et al., 2013). Indeed, there are a number of studies demonstrating the ability of vascular plants to prime decomposition of older soil carbon (Hartley et al., 2012). *Calluna vulgaris* is the most common vascular plant found on grouse moors, due to management encouraging regeneration and germination. There is growing evidence showing that high *Calluna* cover is associated with elevated DOC concentrations in soil water (Vestgarden et al., 2010) and surface drains (Armstrong et al., 2012), as well as studies demonstrating that *Calluna* plants prime decomposition of ancient carbon (Walker et al., 2016). However, much of the organic matter, in both the peat (Bosatta & Ågren, 1999) and the charcoal created from burning (Woolf et al., 2010), is assumed to be stable and recalcitrant meaning it is unlikely that *Calluna* plants themselves greatly affect it. However, ericoid mycorrhizas, whether in association with *Calluna* roots or not, may utilise their saprotrophic abilities to break down either peat or charcoal, or both, and therefore be at least partially responsible for the observed increase in DOC production and water colour over recent decades linked to climate (Worrall et al., 2004), grouse moor management (Clutterbuck & Yallop, 2010) and heather vegetation (Armstrong et al., 2012).

Additionally, given the evidence that *Calluna* plants prime decomposition of ancient carbon (Walker et al., 2016), it may not be solely bacteria which prime decomposition, but also ericoid mycorrhizas, and this additional decomposition will cause an increase in soil respiration and hence CO_2 emissions.

The use of carbon isotopes has been instrumental in the partitioning of respiration derived from (autotrophic) carbon assimilated by photosynthesis and (heterotrophic) respiration derived from decomposition, which may have been stimulated by plant addition (Hahn et al., 2006). As peat accumulates over millennia, the basal peat is relatively depleted in radiocarbon (^{14}C) compared to the upper layers (Borren et al., 2004) due to the exponential decay of ^{14}C. This provides an opportunity to use mass balance approaches to partition respiration into recently derived (modern) CO_2, usually assumed to be that from plants, and older CO_2 from peat decomposition processes (Hardie et al., 2009; Walker et al., 2016). Moreover, the age difference in carbon compounds also allows assessing the impact of addition of (modern) charcoal from heather burning on carbon fluxes. Therefore, a pot experiment using ^{14}C old peat and contemporary charcoal was set up to enable the use of the radiocarbon method to establish whether the presence of charcoal and ericoid fungi affected decomposition, and hence DOC, CO_2 and CH_4 production, specifically considering the possible decomposition of charcoal by ericoid fungi and their impact on old organic matter in the peat. We assessed three hypotheses:

1) Ericoid mycorrhizas result in enhanced decomposition of old peat, particularly when in symbiosis with *Calluna*.
2) Ericoid mycorrhizal fungi can decompose charcoal, particularly when in symbiosis with *Calluna*.
3) The addition of charcoal suppresses decomposition and reduces the DOC concentration in the peat water.

Fig. 109 Site location of the source for the old peat from the Peak District (**left**) used to test mycorrhizal, heather and charcoal effects on decomposition. Charcoal was prepared by burning heather plants from Whitendale (**2nd and 3rd from left**). Heather plants were grown from seed and from cuttings on sterile sand (**2nd from right**) to obtain non-mycorrhizal heather plants which were potted up through 1 µm mesh and inoculated with ericoid mycorrhizal cultures (**far right**).

Methods

A pot experiment was set up to enable manipulation of the different ecosystem carbon flux components and to monitor CO_2 and CH_4 fluxes and pore water DOC concentrations. More details are provided in Appendix 14. This was coupled with radiocarbon (^{14}C) measurements to partition the effects of ericoid fungi, *Calluna* plants and charcoal on DOC, CO_2 and CH_4 fluxes. Peat was collected between 1 and 2 m from the face of a peat bank in the Peak District (**Figure 109** above), which was easily accessible and expected to have a minimum age of 3,000 years based on evidence from Tallis (1991). This provided a strong age contrast against any recently produced carbon from plants. Peat was sterilised by gamma-irradiation and used to set up experimental pots containing sterilised peat. For the pot setup (**Figure 109**), charcoal was produced by burning mature heather plants, cut from a 5 x 1 m area at Whitendale, which produced 88 g or 17.6 g m^{-2} of charcoal. *Calluna* plants were grown in sterilised sand from sterilised seeds and cuttings without any fungal inoculum. The components were then included in a 1 µm mesh sealed pot (to prevent spores from entering) after pre-treatment measurements. Forty eight 25 cm diameter and 22 cm tall sterilised PVC pots with mesh sealed drainage holes were filled with peat and placed on saucers, which were filled and maintained with deionised water (pH adjusted to 3.6). This simulated a WTD in the pots of -18 cm, which was chosen to represent a typical summer WTD (see **Figure 44**). A Rhizon soil moisture sampler (pore size 0.15 µm, Van Walt) was inserted through a hole just above the WTD.

The pre-treatment measurements (see **Table A14.1** in Appendix 14) were used to partition the pots into four blocks, each with 12 plants. Treatments (12) were administered between 6th March and 14th April 2015 and randomly allocated to one pot per block, so there were four pot replicates per treatment. Due to its extensive use in laboratory trials, *Hymenoscyphus ericae* (Read) Korf and Kernam (Strain He 101; supplied by Prof. JR Leake, University of Sheffield) was used as a single mycorrhizal fungus and a wedge of non-sterile peat (from Mossdale) was used to introduce a mixed natural fungal and microbial community. Each pot received either *H. ericae* fungal culture (H), a wedge of non-sterile peat (M) or sterile agar (-), and either *Calluna* plants (C), or not (-), and charcoal (B), or not (-). Plants were planted through a re-sealed slot at the top mesh covering the pot (**Figure 109**). This produced a fully crossed replicated design of 12 treatments with treatment codes listed in **Table 24**. For the pots assigned burnt material, 3.5 g (± 0.012 g) of charcoal was added and evenly mixed into the top 5 cm; this was the equivalent to the total amount of about 4-5 burns.

Table 24 Codes used for the 12 pot treatments and the components of each pot treatment.

Treatment Code	Treatment Components
---	Peat only (and sterile agar)
-C-	*Calluna* plants (and sterile agar)
--B	Burnt material/charcoal (and sterile agar)
-CB	*Calluna* plants and burnt material/charcoal (and sterile agar)
H--	*H. ericae* fungal culture
HC-	*H. ericae* fungal culture and *Calluna* plants
H-B	*H. ericae* fungal culture and burnt material/charcoal
HCB	*H. ericae* fungal culture, *Calluna* plants and burnt material/charcoal
M--	Non-sterile peat wedges
MC-	Non-sterile peat wedges and *Calluna* plants
M-B	Non-sterile peat wedges and burnt material/charcoal
MCB	Non-sterile peat wedges, *Calluna* plants and burnt material/charcoal

Five sets of water samples were collected from the Rhizon samplers in all pots post-treatment over three months. The CO_2 and CH_4 fluxes were measured in parallel five times over three months. Soil moisture content was monitored and maintained at 75% (± 5%) by regular weighing of pots and watering with pH-adjusted (pH 3.6) deionised water. Water samples were analysed for DOC (LiquiTOC, Elementar Analysensysteme) and for UV spectra (Abs254, Abs400, Abs465 and Abs665) using an ultra-violet spectrophotometer (Lambda 25 UV-Vis spectrophotometer, PerkinElmer) and gas fluxes of CO_2 (LiCor 8100) and CH_4 (UGGA; Model 915-0011, Los Gatos Research) were measured. Abs254 was divided by the weight-adjusted (see below) DOC concentration to obtain specific ultra-violet absorbence ($SUVA_{254}$) values and expressed in L mg^{-1} C m^{-1} kg^{-1} dry soil. Water colour was expressed in Hazen units by multiplying Abs400 by 12, following Watts et al. (2001). The relative proportions of fulvic to humic acids were expressed as E4/E6 ratios (Thurman, 1985) by dividing Abs465 by Abs665.

Sampling of ^{14}C used custom-built 20 cm long and 10 cm diameter uPVC chambers which were fitted over the central collars in the pots by means of thick rubber bands to create an airtight seal (**Figure 110**). An established molecular sieve sampling system (see Hardie et al. (2005) and Garnett & Murray (2013) for full details) was used to collect samples for ^{14}C analysis (**Figure 110**; right). The DOC samples for ^{14}C analysis were collected by attaching acid-washed 50 ml luer-lock syringes to the pot Rhizon samplers (**Figure 110**; left). The molecular sieve cartridges and DOC samples, along with three peat samples and a charcoal sample which were obtained before treatment addition, were sent to the NERC Radiocarbon Facility (East Kilbride, UK) for analysis (NRCF010001; allocation number 1841.1014). A multi-component isotope mass balance approach, based on the two- and three-component mass balance approaches used by Hardie et al. (2009), was employed to derive the fluxes and isotopic concentrations of each pot component.

Fig. 110 Fully assembled sealed pots with attached syringes for peat pore water collection with Rhizon samplers (**left**) and during ^{14}C gas collection (15/10/15) using a molecular sieve sampling system (**right**).

Results

The DOC concentrations in the soil pore water of the pots averaged 99.2 mg C L^{-1} kg^{-1} dry soil before treatment addition, which was significantly higher ($F_{1,3}$ = 173.77, p< 0.001) than the post-treatment average of 20.7 mg C L^{-1} kg^{-1} dry soil. However, the DOC concentrations did not differ significantly between treatments either pre- or post-treatment ($F_{11,34}$ = 0.78, p = 0.66). Similarly, there was no significant interaction in $SUVA_{254}$ between treatment and period ($F_{11,46}$ = 0.85, p = 0.59;). In contrast to DOC, $SUVA_{254}$ was significantly higher post-treatment ($F_{1,32}$ = 422.28, p< 0.0001), with an average of 7.2 L mg^{-1} C m^{-1} kg^{-1} dry soil compared to 2.0 L mg^{-1} C m^{-1} kg^{-1} dry soil pre-treatment. The colour of the water samples ranged from almost clear to tea-brown, with an average of 270 Hazen units. The E4/E6 ratios were between 0.99 and 5.16 with a mean of 2.44. Although both Hazen units and E4/E6 ratios were slightly higher post-treatment, these increases were not significant ($F_{1,3}$ = 1.72, p = 0.28 and $F_{1,3}$ = 5.52, p = 0.10, respectively). There was also no significant interaction between the periods and the treatments ($F_{11,60}$ = 0.68, p = 0.75 and $F_{11,48}$ = 0.53, p = 0.87, respectively). For a detailed summary of the soil pore water analysis see **Table A14.3** in Appendix 14.

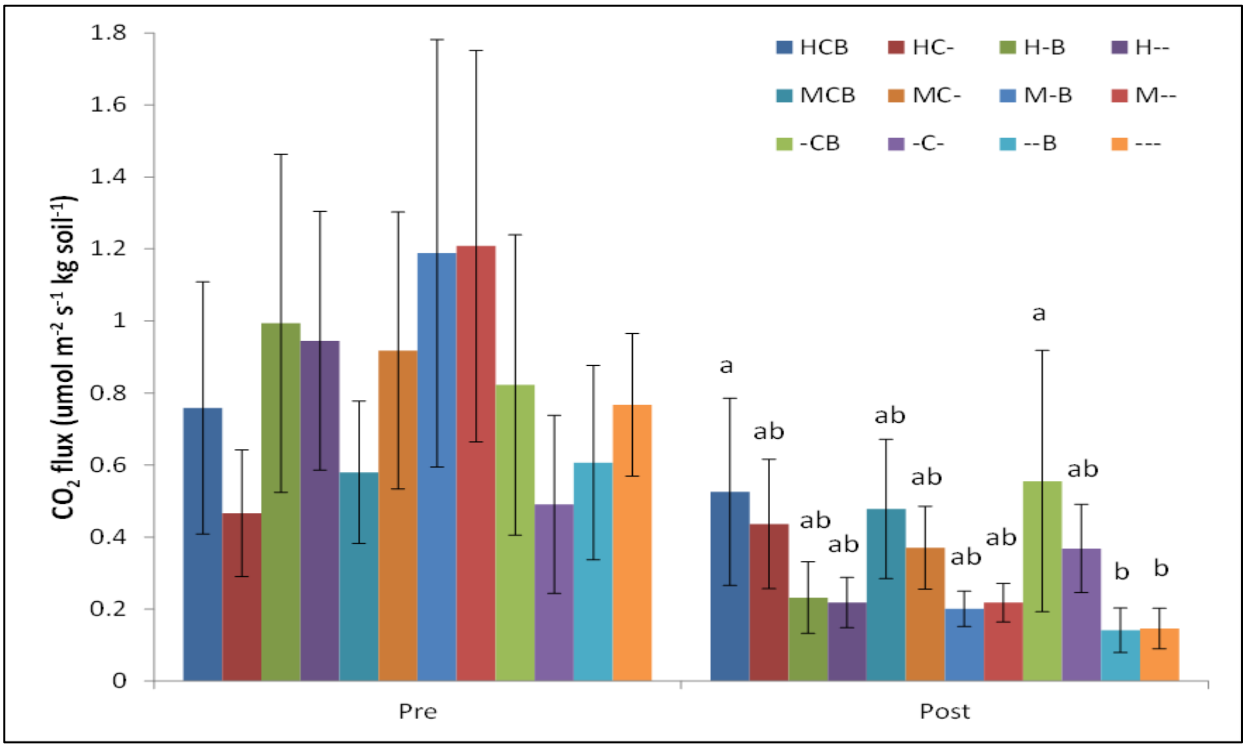


Fig. 111 Pot treatment averages of the CO_2 fluxes (± 95% confidence interval; CI) in the pre- and post-treatment periods. Treatment codes are explained in **Table 24**. Different letters indicate significant differences between managements in the post-management period; pre-treatment differences were not significant.

As for DOC concentrations, CO_2 fluxes from the pots reduced significantly following treatment addition ($F_{1, 370}$ = 87.77, p< 0.0001; **Figure 111** above). There was also a significant interaction between the treatment period and the treatments ($F_{11,370}$ = 3.54, p< 0.0001). When only the post-treatment CO_2 fluxes were analysed, there was a significant difference between treatments ($F_{11,33}$ = 2.69, p = 0.0138). Fluxes from HCB and -CB pots were significantly higher than fluxes from --B and --- pots (p< 0.04 for all), likely reflecting increased soil fluxes due to the presence of *Calluna* roots.

The CH_4 fluxes ranged from -0.590 to 0.691 nmol m^{-2} s^{-1} per kg dry soil, although many pots had flux values close to zero. The average post-treatment CH_4 flux (-0.009 nmol m^{-2} s^{-1} per kg dry soil) was slightly lower than the average pre-treatment flux (-0.002 nmol m^{-2} s^{-1} per kg dry soil) but there was no significant difference between the two time periods ($F_{1,372}$ = 0.89, p = 0.35). There was also no significant interaction between treatment and time period ($F_{11,372}$ = 1.14, p = 0.33).

The CO_2 fluxes differed significantly between the five treatments chosen for radiocarbon analysis ($F_{4,7}$ = 6.90, p = 0.0130), with the pots containing *Calluna* plants producing significantly higher fluxes than the --- and --B pots (p< 0.016 for all; **Figure 112**; left). The HCB pots also produced significantly higher fluxes than the H-- pots (p = 0.013). Similarly, the $^{14}CO_2$ content differed between treatments ($F_{4,9}$ = 11.73, p = 0.0013), with pots containing *Calluna* showing significantly higher enrichment (p< 0.045 for all). One MCB pot was included in the analysis as a substitute for one HCB pot (due to plant death in the HCB pot) to enabled to consider a set of three replicates of the mycorrhizal plant and charcoal treatment in the statistical analysis. However, it was noted that the pot ecosystem $^{14}CO_2$ content of the MCB pot was 10 %Modern lower than that of the two HCB pots (74.79 %Modern compared to 84.68 %Modern), equating to a difference of 1,000 years. Appendix 14 (**Table A14.4**) provides a complete list of individual pot ^{14}C contents, publication codes and sample types.

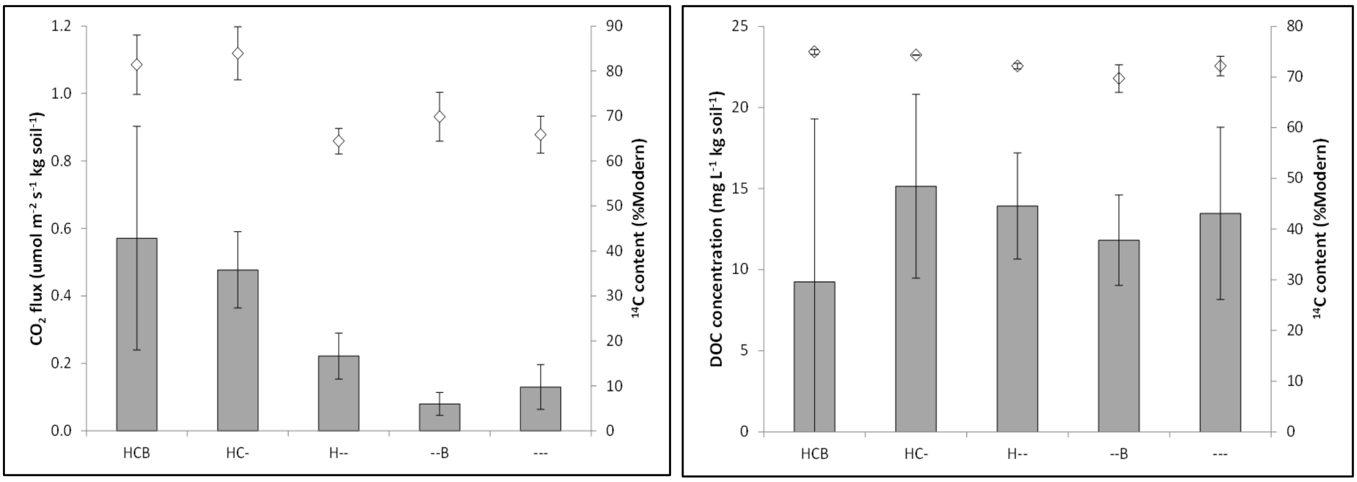


Fig. 112 Pot treatment averages of (**left**) the CO_2 fluxes (grey bars), measured the day before radiocarbon sampling, and the ^{14}C content of the CO_2 (white diamonds), and (**right**) DOC concentrations (grey bars), measured in the water samples taken for radiocarbon sampling, and the ^{14}C content of the DOC (white diamonds). Error bars are 95% confidence interval. Treatment codes are explained in **Table 24**. Note the different scales on the axes.

The presence of charcoal reduced the DOC concentrations by a third when in combination with *Calluna* and the single mycorrhizal fungi species (**Figure 112**; right), although there was no significant difference in DOC concentration between treatments ($F_{4,9}$ =0.50, p = 0.73). The DOC concentrations in the HCB pots contained the highest ^{14}C content. Despite there being a substantially lower effect of treatment for the ^{14}C content of DOC than of CO_2, the pot ecosystem $DO^{14}C$ content also differed significantly between treatments ($F_{4,9}$ = 7.66, p = 0.0057). HCB pots showed significantly more ^{14}C enrichment than --B, H-- and --- pots (p< 0.035 for all) and HC- pots had a higher DOC ^{14}C content than --B pots (p< 0.001). There was however no difference in the $DO^{14}C$ content between the MCB pot and the two HCB pots (74.94 %Modern and 74.92 %Modern, respectively (see **Table A14.4** in Appendix 14).

The *Calluna*-associated CO_2 flux calculated by the mass balance approach showed that *Calluna* contributed 45% of the flux in the HCB pots and 54% in the HC- pots. Notably, 41% of the CO_2 flux of the H-- pots was attributable to *H. ericae* fungi, despite the fungal DOC contribution to the H-- pots being only 3%. The *Calluna* contribution to the DOC concentration was much smaller, accounting for 13% of the DOC produced by the HCB pots, assuming that the lower DOC concentration in these pots was entirely caused by the charcoal and 8% of the DOC produced by the HC- pots. As for *Calluna*, the *H. ericae* fungal contributions to the DOC concentrations were much smaller than to the CO_2 fluxes, comprising 5% (again assuming concentration reduction by charcoal) and 3% of the HCB and HC- pot DOC concentrations, respectively, and 16% and 19% of the CO_2 fluxes, respectively.

Due to the combination of treatments chosen for radiocarbon analysis, the charcoal component flux contributions could be calculated in two different ways (see Appendix 14). When the --B and --- pots were used, the contributions of charcoal to CO_2 flux and DOC concentration were negative, reducing the values by 64% and 12%, respectively, relative to the --- pots. However, when calculated using the HCB and HC- pots, the charcoal component increased the CO_2 flux by 16%, although the presence of charcoal reduced the DOC concentration by 39%.

The respired CO_2 from the peat was over 300 years older than the bulk peat sample (which had an age of 3,021 ± 91 years old, i.e. was 68.66 ± 0.78 %Modern) whereas the DOC was almost 400 years younger (**Table 25**). The mass balance calculations revealed that *Calluna* plant respiration was modern (100.85 %Modern) and *Calluna*-attributable DOC was only 73 years BP (99.10 %Modern). Strikingly, both the *H. ericae* fungal respiration (62.37 %Modern) and the DOC portion attributable to the fungus (71.24 %Modern) were older than the respective peat components by 436 and 101 years, respectively (**Table 25**). Both values indicated strong positive priming by the presence of *H. ericae* of decomposition of old carbon. The bulk charcoal sample was modern, with an average age of about 8 years old (approximately from the year 2009; 104.95 ± 0.48 %Modern; 40.9% C content by weight).

Due to the combination of dated treatments, the charcoal component ages could be calculated in two different ways. Using the --B and --- pots (giving the effect of charcoal addition to peat in the absence of vegetation) produced ages of 4,172 and 865 years BP (59.49 and 89.79 %Modern) for CO_2 and DOC respectively. In contrast, when the HCB and HC- pots were used (i.e. in the presence of *Calluna* and *H. ericae*), the charcoal CO_2 and DOC components were calculated as 3,042 and 2,493 years BP (68.48 and 73.32 %Modern) respectively. As all calculated charcoal CO_2 and DOC ages were substantially older than the charcoal sample (contemporary: 104.95 %Modern), they could not be derived directly from the charcoal and therefore should be considered as charcoal-induced component ages and fluxes. Additionally, since only the latter value for the CO_2 age was calculated using a positive flux (and both DOC charcoal-induced component concentrations were based on negative DOC concentrations), these ages should be treated with caution.

Table 25 Radiocarbon (^{14}C) ages in years BP and the associated CO_2 and DOC contributions for each treatment component calculated using a mass balance approach. The first values for the charcoal-induced component were calculated using --B and --- pots and the second using HCB and HC- pots (see Appendix 14 for more details). All values are derived from averages of three pots apart from the first charcoal-induced age and flux which only used two pots in the --B average.

Flux component	CO_2 age (years BP)	DOC age (years BP)	CO_2 flux (μmol m^{-2} s^{-1} kg dry soil^{-1})	DOC concentration (mgC L^{-1} kg dry soil^{-1})
Peat	3356	2623	0.13	13.46
H. ericae fungus	3792	2724	0.09	0.46
Calluna plants	Modern	73	0.26	1.23
Charcoal	4172 or 3042	865 or 2493	-0.05 or 0.09	-1.65 or -5.91

Discussion

The results from this study only supported the first hypothesis: (1) ericoid mycorrhizas results in enhanced decomposition of old peat, particularly when in symbiosis with *Calluna*. These are the first data to suggest that ericoid mycorrhizal fungi break down old and assumed to be recalcitrant C compounds in peat and release this C both as DOC and CO_2. This was evidenced by the ages of the ericoid component of CO_2 and DOC release being older than the base peat component CO_2 and DOC ages. Although the fungal contributions to soil DOC were less than 5%, the contributions to the CO_2 flux were up to 20%, and the function of ericoid fungi in decomposition of recalcitrant organic matter should clearly be considered further in the context of peatland C cycling. Despite the (heterotrophic) decomposition-related component strongly influencing the age of the CO_2 and DOC, the *Calluna* plants had a far greater influence on the quantity of CO_2 and DOC produced, partially due directly to the (autotrophic) plant inputs but also because these inputs seemed to prime the microbial decomposition of older organic matter. In light of the expectation that climate change will probably increase plant growth, that greater plant growth is associated with a larger priming effect (Hartley et al., 2012) and that *Calluna* appears to exhibit greater priming effects than other peatland vegetation (Walker et al., 2016), these results demonstrate that *Calluna* may threaten the longevity of the peatland C store. However, there was no impact of ericoid mycorrhizas on methane fluxes.

Over the past 50 years, there has been an increase in the amount of DOC and in the colour of the water leaving peat-covered catchments (Monteith et al., 2007), which is of concern to policy. Bragazza et al. (2013) suggested that “the quantity and quality of DOC might be controlled by vascular plants through a greater rhizodeposition of labile C compounds”. There is also mounting evidence that both warming (Dorrepaal et al., 2009) and certain types of vegetation (Walker et al., 2016) can cause the liberation of old carbon from peatlands which, amongst other pressures such as drainage and burning, casts doubt upon the longevity of peat as a stable long-term C store. Lu et al. (2014) found negative effects on DOC concentrations by biochar. However, the other two hypotheses of (2) ericoid mycorrhizal fungi can decompose charcoal, particularly when in symbiosis with *Calluna*, and (3) the addition of charcoal suppresses decomposition and reduces the concentration of DOC in the peat pore water could not be confirmed by this study.

The small reductions in DOC post-treatment DOC concentrations observed here when mycorrhizal plants also had charcoal addition (**Figure 112**) were not significant. The reason for such a charcoal effect is unknown, but it could be due to adsorption of DOC to the charcoal, or the charcoal causing changes in pH and nutrient availability which may have altered plant growth or mycorrhizal activity, resulting in differences in DOC release. However, burning may counteract some of the mycorrhizal induced decomposition since charcoal made from *Calluna* plants appeared to reduce DOC concentrations (but only for the single ericoid species in symbiosis with *Calluna*) by over a third (see **Figure 112**; right) whilst increasing CO_2 emissions by only about a fifth (**Figure 112**; left). Additionally, burning has been shown to reduce *Calluna* cover and fungal biomass but to increase plant C uptake and C transfer to soil microbes (Ward et al., 2012). However, at least some of the apparent charcoal effect was likely to be due to the charcoal and ash changing some of the physico-chemical peat properties. This raises the question as to whether the apparent effects of charcoal addition would be the same under a higher WTD and for how long. Further investigation into charcoal and its interaction with vegetation and microbes, particularly ericoid fungi, on peatland C dynamics are needed to resolve these uncertainties.

In summary, the controlled studies of peat decomposition processes described in Sections 4.4.2, 4.4.3 and 4.4.4 have shown that:

- Q_{10} values for decomposition rates were significantly higher on mown plots with brash left than on burnt plots. The mown plots also showed a stronger effect of soil moisture on decomposition rates.
- The mown plots had higher lignin content in surface peat, consistent with input from the remaining brash. Burnt plots had a higher charcoal content in surface peat.
- The age of the C released as both CO_2 and DOC that was associated with the presence of mycorrhizal fungi was older than that from peat only. This suggests that the mycorrhizal fungi associated with heather are able to break down very old peat carbon, potentially threatening the longevity of the peatland C store.
- This previously unrecognised process may provide a further mechanism to explain the increase in DOC concentrations and water colour from peat catchments in recent decades.

4.5 *Cranefly emergence, abundance and impacts on birds*

Craneflies are a keystone insect species in blanket bog ecosystems, dominating the soil fauna's abundance and biomass (Coulson, 1988). They can constitute a major dietary component for peatland breeding birds (Buchanan et al., 2006; Pearce-Higgins, 2010) and higher availability of adult craneflies has been associated with higher chick growth and survival (Park et al., 2001; Pearce-Higgins & Yalden, 2004). The potential management and climatic impacts in upland systems, specifically considering moisture impacts on craneflies (*Diptera*: *Tipulidae*), are particularly relevant to populations of upland birds as shown by Carroll et. al (2011). Therefore, two survey methods were performed at all three sites assessing climate and management impacts on cranefly emergence (using traps) and abundance (along transects). Findings were then related to cranefly mediated impacts on key upland bird species. This reflected the practitioner focus of the overall project within the Project Advisory Group (including land owners and users) in relation to impacts of alternative heather management on blanket bogs on bird populations, including the economically important red grouse as well as localised species of conservation status such as golden plover and dunlin. Three model approaches were used in order to predict impacts on bird abundance. One approach used relationships developed by Carroll et al. (2011) of cranefly trap emergence, peat (soil) moisture and bird abundance (i.e. golden plover, dunlin and red grouse); the other two applied empirical relationships of cranefly abundance across transects with breeding success and chick survival of golden plover (Pearce-Higgins & Yalden, 2004) and the number of breeding pairs also including vegetation height (Douglas & Pearce-Higgins, 2014). Novel aspects of the work included assessing plot-level and catchment-level management impacts on upland birds, reflecting both craneflies and environmental impacts, which enabled upscaling in space and time including considering future scenarios of management and climate.

Summary of methods

Cranefly emergence was assessed at all plots following the method developed by Carroll et al. (2011) i.e. sticky traps are attached to the inside of upturned baskets pegged to the ground which trap the craneflies that emerge (see **Figure 113**). Each monitoring (treatment) and slope plot had one trap installed (see Appendix 10 for statistical information). However, to achieve a balanced sampling design, burnt plots had an additional four traps installed on the *Sphagnum* addition sub-plots. This resulted in five traps in each of the four plot blocks, plus six traps in each of the three slope blocks, in each burnt or mown sub-catchment, giving a total of 38 (20 plus 18) traps in each sub-catchment. Traps were deployed in 2014, 2015 and 2016. Traps were deployed each year during April until July, covering the main cranefly emergence period. Traps were visited every 3-4 weeks, the numbers of craneflies trapped were counted, and craneflies were then removed or sticky traps replaced. Numbers for each visit were summed to give the total cranefly abundance for each trap for each year. At the start of the 2015 and 2016 trapping periods, traps were moved adjacent to their previous position on an area of similar ground cover. Soil moisture (top 8 cm; HH2, SM200, Delta T) and soil and air temperatures were also recorded at each trap location on each visit. In addition, cranefly abundance (flying and sitting) was assessed on 10 m transects (4 metre wide) to and from each emergence monitoring area (i.e. per sub-catchment consisting of four blocks containing the five 5 x 5 m plots and the three slope areas).

Fig. 113 Example of a sticky trap to collect craneflies during the bird breeding season (mainly April-July) according to Carroll *et al*. (2011). Emergence traps (baskets of L41 cm × W28 cm × H17 cm; far left) covered on the inside with sticky tape were pegged to the ground (**centre left**) and trapped craneflies were counted at regular intervals by lifting (**centre right**) the basket (and replacing tape if needed). To and from each trap a 10 m transect (**far right**) was also assessed for cranefly abundance (flying or sitting). Peat surface moisture and soil and air temperatures were also recorded (see **middle pictures**).

4.5.1 Relationship between soil moisture and cranefly trap emergence

Combining data from all sites and years, there was a significant (p< 0.001) positive wedge-shaped relationship (i.e. abundances were low at relatively dry sites but could be high or low at relatively wet sites) between soil moisture and cranefly emergence (**Figure 114**; **Table 26**), with an increase in trap abundance particularly noticeable above 85% soil moisture. The overall numbers as well as the relationship was very similar to those reported previously by Carrol et al. (2011). However, the peak abundance was observed around 95%, with levels decreasing nearer 100% and below 80%, indicating an ~85-95% optimum soil moisture range supporting large cranefly emergence (suggesting possible impacts on larval survival rates; Coulson, 1962). Appendix 10 provides a complete graphical and statistical output for these data.

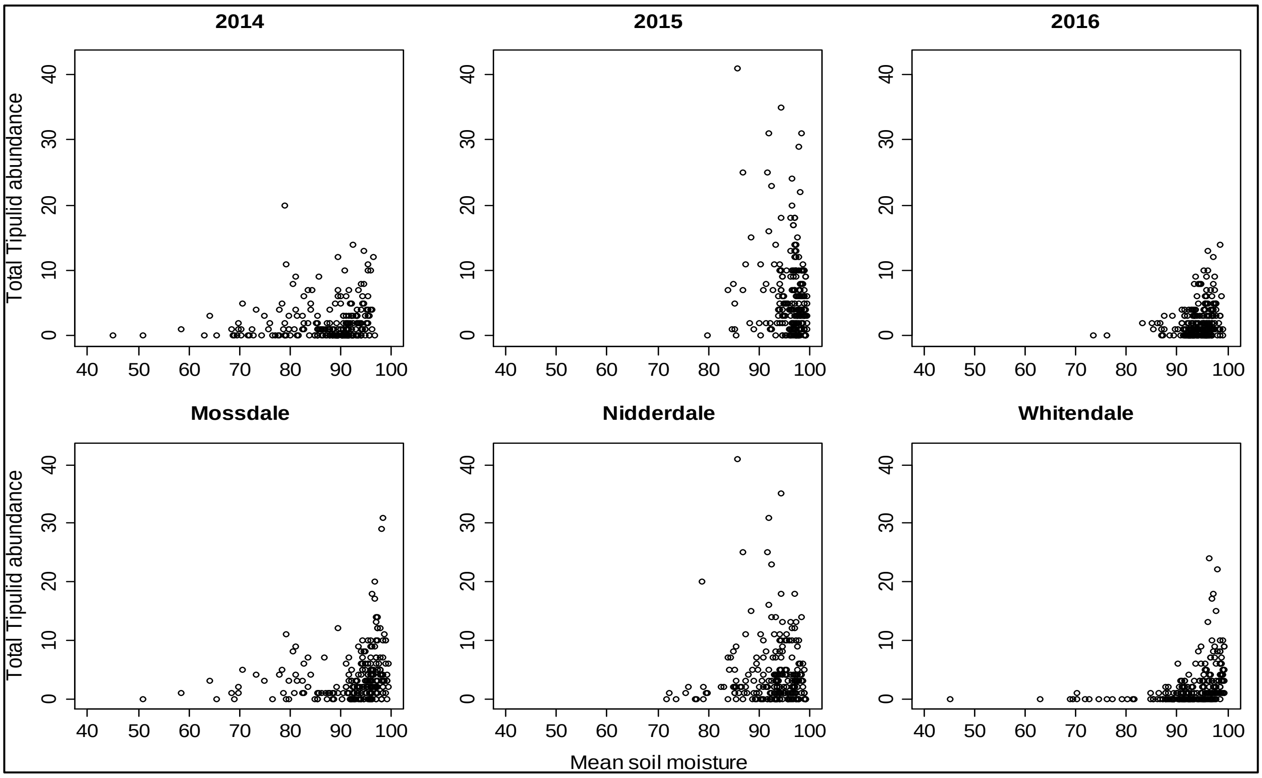


Fig. 114 Relationship between cranefly (Tipulid) emergence (abundance) in sticky traps (totals per trap) and soil moisture (%) at all cranefly traps for each year with sites combined (**top row**) and each site with years combined (**bottom row**).

Table 26 Results from GLMMs with negative binomial error and log link, describing cranefly trap abundance as a function of mean soil moisture. An offset of ln (number of trapping days) was entered to account for variation in trapping period between sites and years. Random effects were trap nested in block, block nested in sub-catchment, and sub-catchment nested in site and year. The number of data points differed between sites and years due to the loss of some traps which were blown away or trampled by livestock. Bold font highlights the moisture slope was significantly different from zero at P< 0.05. Coefficients represent slopes of relationship. All models include plot and slope data, apart from those shaded in grey which include only plot data i.e. locations in which experimental management had taken place.

Data subset	Significance of random effects					Fixed effects				
	Year	Site	Sub-C.	Block	Trap	Coeff	SE	DF	t-value	P-value
All years	**<0.001**	**0.635**	**0.351**	**0.667**	**1.129**	**0.035**	**0.009**	**544**	**3.779**	**<0.001**
2014	**-**	**0.775**	**0.767**	**0.582**	**-**	**0.022**	**0.011**	**182**	**2.050**	**0.042**
2015	-	<0.001	<0.001	0.568	-	-0.030	0.023	175	-1.323	0.188
2016	**-**	**0.131**	**0.224**	**0.476**	**-**	**0.104**	**0.028**	**185**	**3.711**	**<0.001**
Mossdale	**<0.001**	**-**	**<0.001**	**0.615**	**0.876**	**0.022**	**0.010**	**185**	**2.201**	**0.029**
Nidderdale				(no model convergence)						
Whitendale	**<0.001**	**-**	**0.491**	**0.411**	**<0.001**	**0.144**	**0.036**	**174**	**4.005**	**<0.001**
Burnt	<0.001	0.598		0.559	<0.001	0.022	0.016	270	1.429	0.154
Burnt	0.431	1.168		0.418	0.869	0.005	0.017	142	0.304	0.762
Mown	**<0.001**	**0.432**		**0.513**	**1.098**	**0.030**	**0.015**	**273**	**2.027**	**0.044**
Mown	<0.001	0.369		0.270	0.991	-0.008	0.021	140	-0.381	0.704

Considering years individually, there was a significant positive relationship between cranefly emergence and soil moisture for 2014 ($p < 0.05$) and 2016 ($p < 0.001$), but not in 2015 ($p = 0.188$), when conditions were very wet (**Figure 114**; **Table 26**). Data from Mossdale and Whitendale showed that the significant positive relationship between cranefly emergence and soil moisture held at the site level but not at Nidderdale due to convergence issues of the statistical model (**Table 26**). However, significant positive relationships were only observed when all datasets (i.e. plot and slope locations) were combined, which was also the case for the mown catchment; when only the plot data, where management had taken place, were used, the effect was non-significant (**Table 26**). However, plot (management) locations only covered a very narrow soil moisture range (as they were predominantly flat areas); the inclusion of the drier slope locations expanded the moisture range and therefore revealed the overall significant relationship, which was of a similar shape (see **Figure A10.3** in Appendix 10) but too narrow to be statistically significant.

4.5.2 Effect of catchment management on soil moisture

Taking all data together, soil moisture was significantly higher ($p < 0.001$) in mown (mean = 93.1%; σ =6.09) than burnt (mean = 91.7%; σ =7.48) catchments (**Figure 115(a)**). When only data from the plots (not slopes) was included in the analysis, soil moisture was significantly and similarly higher ($p = 0.002$) in mown (mean = 94.1%; σ = 5.23) than burnt (mean = 92.6%; σ = 6.37) catchments (**Figure 115(b)**). The difference was also similar when the uncut (DN) treatment was also removed, with significant ($p = 0.004$) soil moisture differences between the mown (mean = 94.1%; σ = 5.29) and burnt (mean = 92.6%; σ = 6.37) sub-catchments (**Figure 115(c)**). Although the mean difference was not very large, the lower soil moisture together with the lower moisture ranges on burnt than on mown catchment plots, corresponds to the observed lower water table depths in burnt compared to mown and uncut plots (see **Table 9**) and lower catchment flow rates from mown areas (see **Table 11**). Notably, drying out at the peat surface can happen quickly on exposed sites despite high water tables, particularly with larger areas of exposed peat after burning compared to mown plots (**Figure 34a,b)**, particularly with brash left (**Figure 35**).

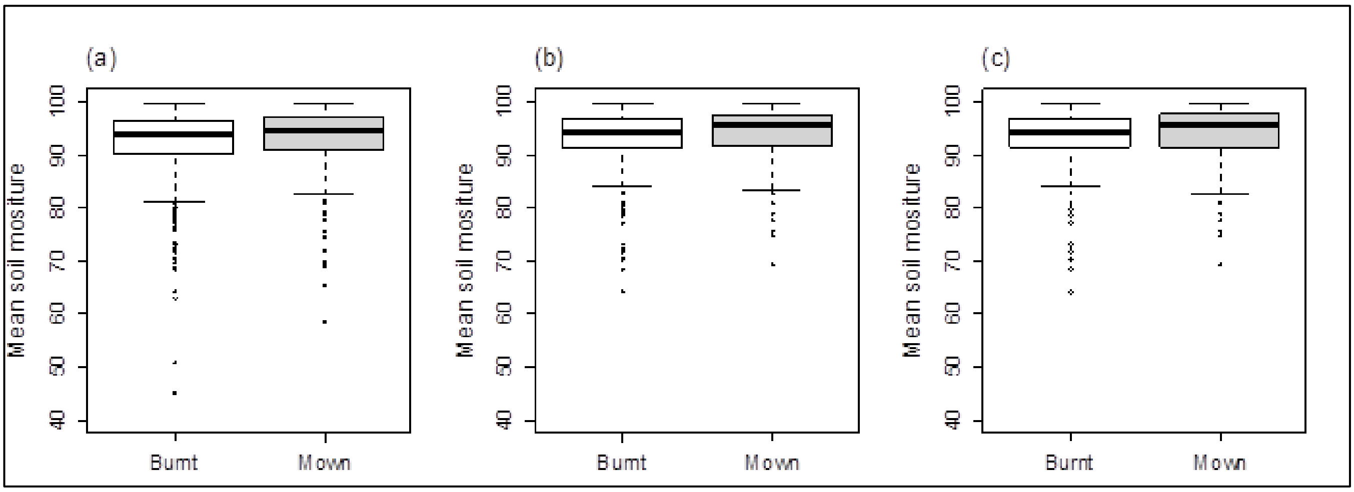


Fig. 115 Comparison of soil moisture (%) in burnt and mown sub-catchments during 2014-16 (using annual means per monitoring location), using (**a**) all data; (**b**) data just from plots (not slopes); and (**c**) also excluding traps in uncut (DN) sub-treatments. Box midline indicates median, box edges indicate interquartile range, whiskers indicate range of data and points indicate data outside 1.5 x the interquartile range.

When considering the data for individual years, soil moisture was significantly higher in mown than burnt sub-catchments in 2015 ($p < 0.001$) and 2016 ($p = 0.003$) but not in 2014 ($p = 0.084$), although the difference was approaching the significance threshold at $p < 0.05$ (**Figure 116**). When data from slope plots was excluded, the effect was the same (2014: $p = 0.088$, 2015: $p = 0.003$; 2016: $p = 0.026$), as it was when also excluding uncut plots (2014: $p = 0.133$; 2015: $p = 0.005$; 2016: $p = 0.012$). The lowest values were always found on burnt areas, suggesting that mowing reduced the extent to which the peat surface dried out during spring and summer

compared to the burnt management, both at the plot and catchment level. For a more detailed statistical output see the Appendix 10.

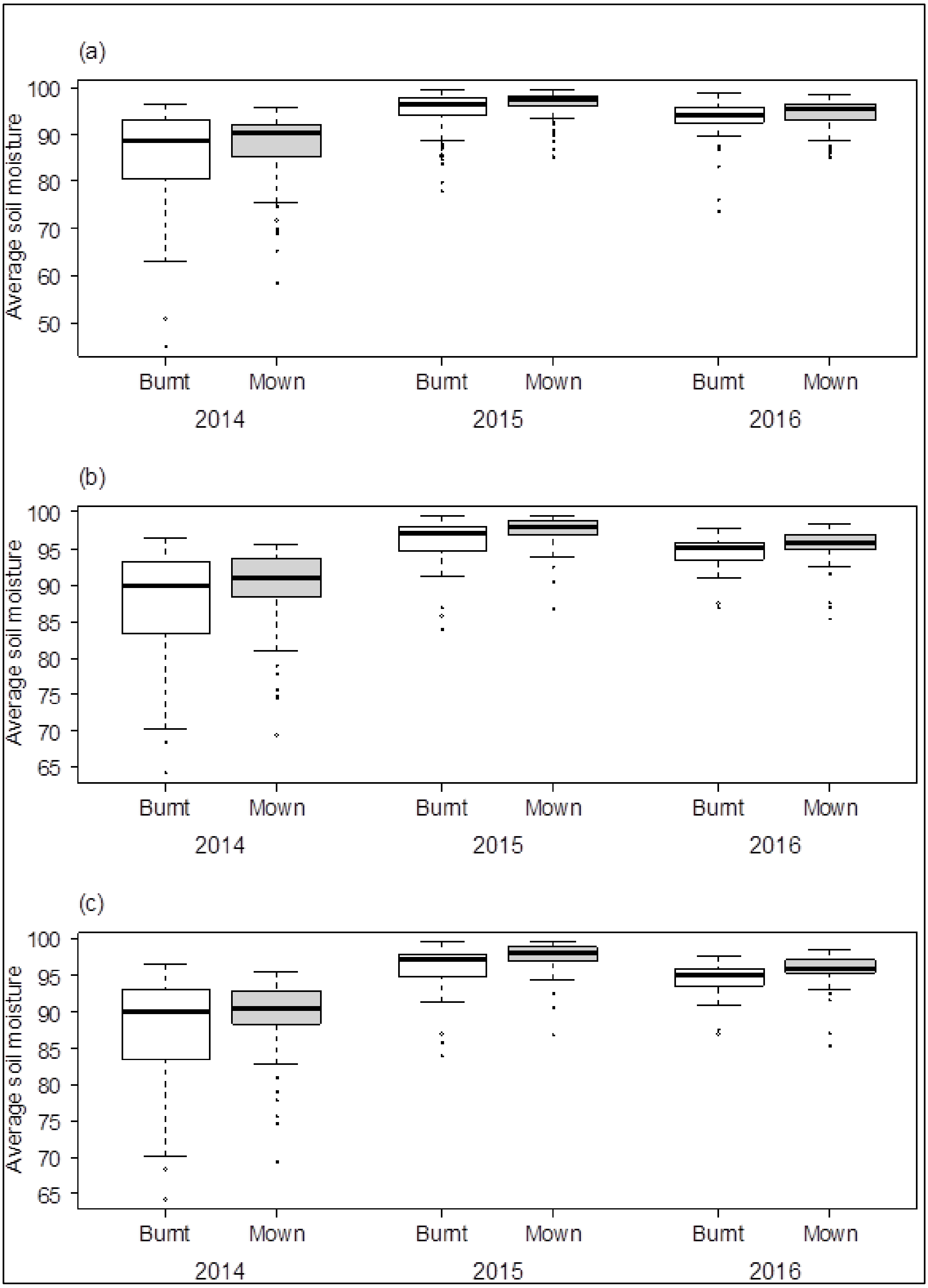


Fig. 116 Comparison of soil moisture (%) in burnt and mown sub-catchments in separate years (using annual means per monitoring location), using (**a**) all data; (**b**) data just from plots (not slopes); (**c**) also excluding traps in uncut (DN) plots. Box midline indicates median, box edges indicate interquartile range, whiskers indicate range of data and points indicate data outside 1.5 x the interquartile range.

Comparing the observed moisture effect at the individual sites (**Figure 117**) indicated very similar soil moisture differences between mown and burnt managements. The difference between drier burnt and wetter mown areas increased, as did the significance levels, when the unmanaged slope plots were excluded. However, none of the differences between burnt and mown areas was significant at Nidderdale.

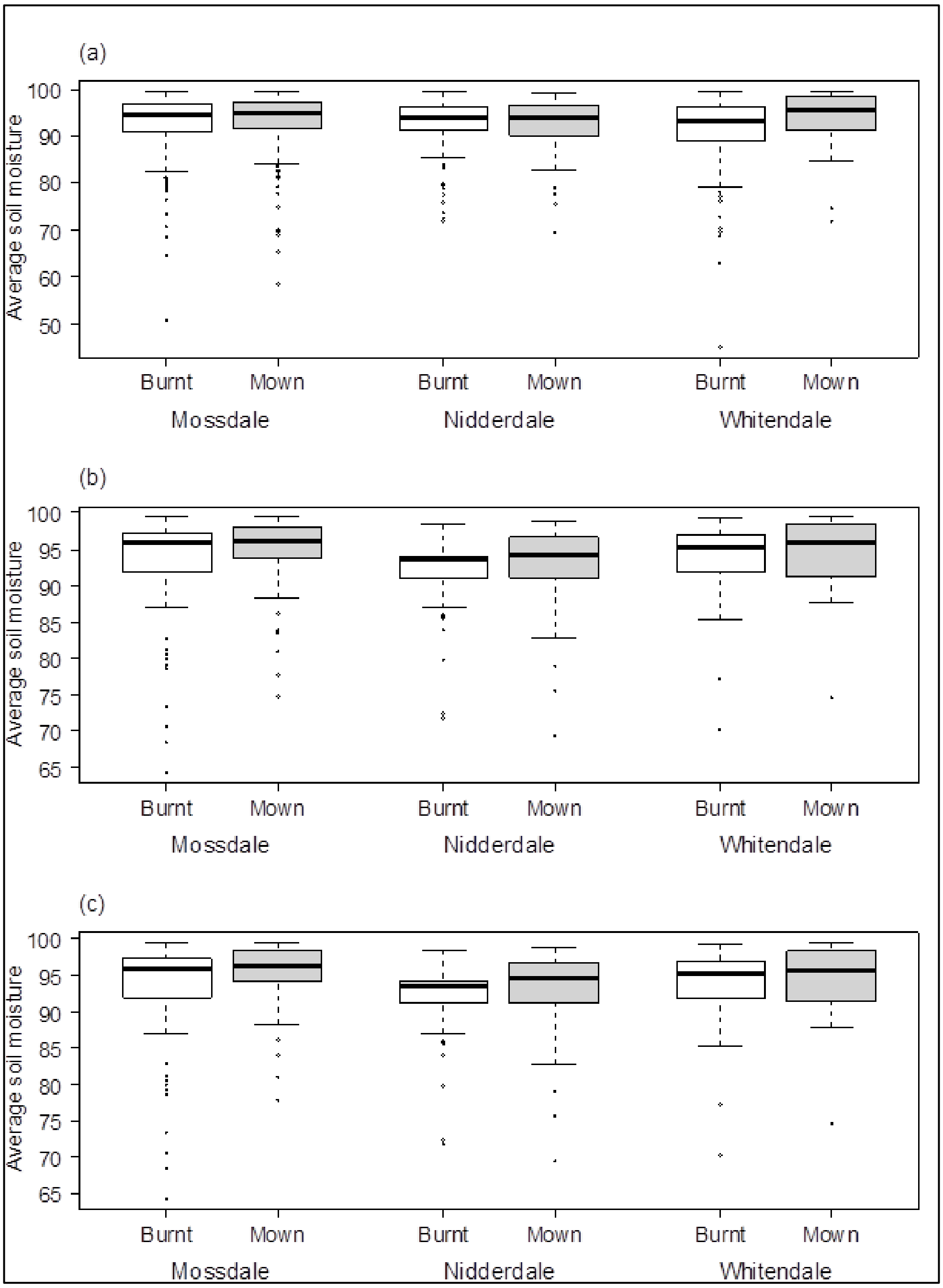


Fig. 117 Comparison of soil moisture (%) in burnt and mown sub-catchments in separate sites (using annual means per monitoring location), using (**a**) all data; (**b**) data just from plots (not slopes); and (**c**) also excluding traps in uncut (DN) plots. Box midline indicates median, box edges indicate interquartile range, whiskers indicate range of data and points indicate data outside 1.5 x the interquartile range.

When comparing different years at the individual sites, the difference in soil moisture revealed similar trends of wetter conditions across mown catchments compared to burnt catchments (see Appendix 10). However, for the wetter years of 2015 and 2016, this resulted in median soil moistures much closer to 100% for the mown than in the burnt catchments, particularly for Mossdale. Although small overall, the reductions in soil moisture in burnt catchments are likely to be ecologically important considering the narrow optimum range of soil moisture (~85-95%) for high cranefly emergence (**Figure 114**).

4.5.3 Effect of catchment management on cranefly trap emergence

Despite these consistent differences in soil moisture between mown and burnt sub-catchments, and the strong relationship between soil moisture and cranefly abundance, when cranefly trap data from all sites and years was combined, there was no significant difference in cranefly trap abundance between burnt (mean = 4.95; σ = 6.30) and mown (mean = 4.13; σ = 4.85) sub-catchments. This was also the case after excluding slopes and uncut plots. Considering each year separately, cranefly emergence was significantly higher ($p < 0.001$) in mown than burnt sub-catchments in 2014 but there was no significant difference in 2015 ($p = 0.268$), and in 2016 cranefly emergence was significantly higher ($p < 0.01$) in burnt than mown sub-catchments (**Figure 118**). Notably, the reduction in cranefly emergence in mown compared with burnt areas also became significant in 2015 when excluding slopes and uncut plots ($p = 0.020$). For detailed statistical information see Appendix 10.

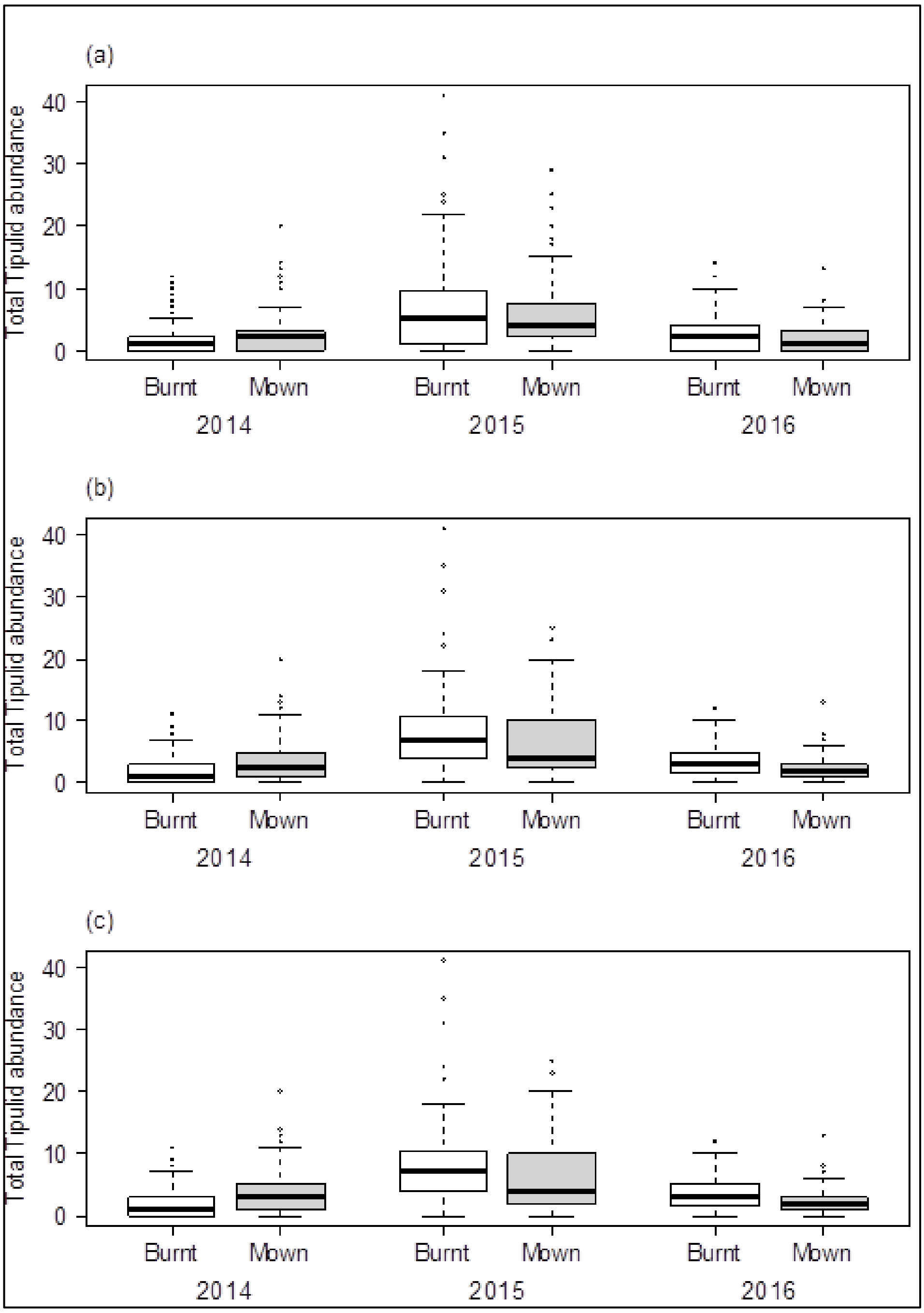


Fig. 118 Comparison of total cranefly (Tipulid) trap emergence (abundance) between burnt and mown sub-catchments for separate years, using (**a**) all data; (**b**) data just from plots (not slopes); and (**c**) also excluding traps in uncut (DN) plots. Box midline indicates median, box edges indicate interquartile range, whiskers indicate range of data and points indicate data outside 1.5 x the interquartile range.

Comparing the different sites across all three years, cranefly trap emergence was lowest at Whitendale, the sites with the highest overall soil moisture range (**Figure 114**), and highest at the driest site Nidderdale. This indicated two contrasting moisture impacts on cranefly emergence. Whilst Whitendale did show lower median but higher interquartile ranges for cranefly emergence in burnt than in mown catchments, the generally wettest (i.e. highest water tables) site Mossdale (**Figure 45**) was also the only site where cranefly abundance was higher ($p< 0.001$) in burnt than mown sub-catchments (**Figure 119**). This result suggests a potential negative impact of soil moisture values above 95%, which are more likely on mown and already very wet sites, on larval cranefly mortality, potentially through drowning. Nidderdale showed an overall lower soil moisture (**Figure 117**) and significantly higher ($p = 0.029$) cranefly abundance under mown management (i.e. indicating buffering against desiccation of larvae in drier site conditions), but only when excluding slope and uncut plots (**Figure 119**).

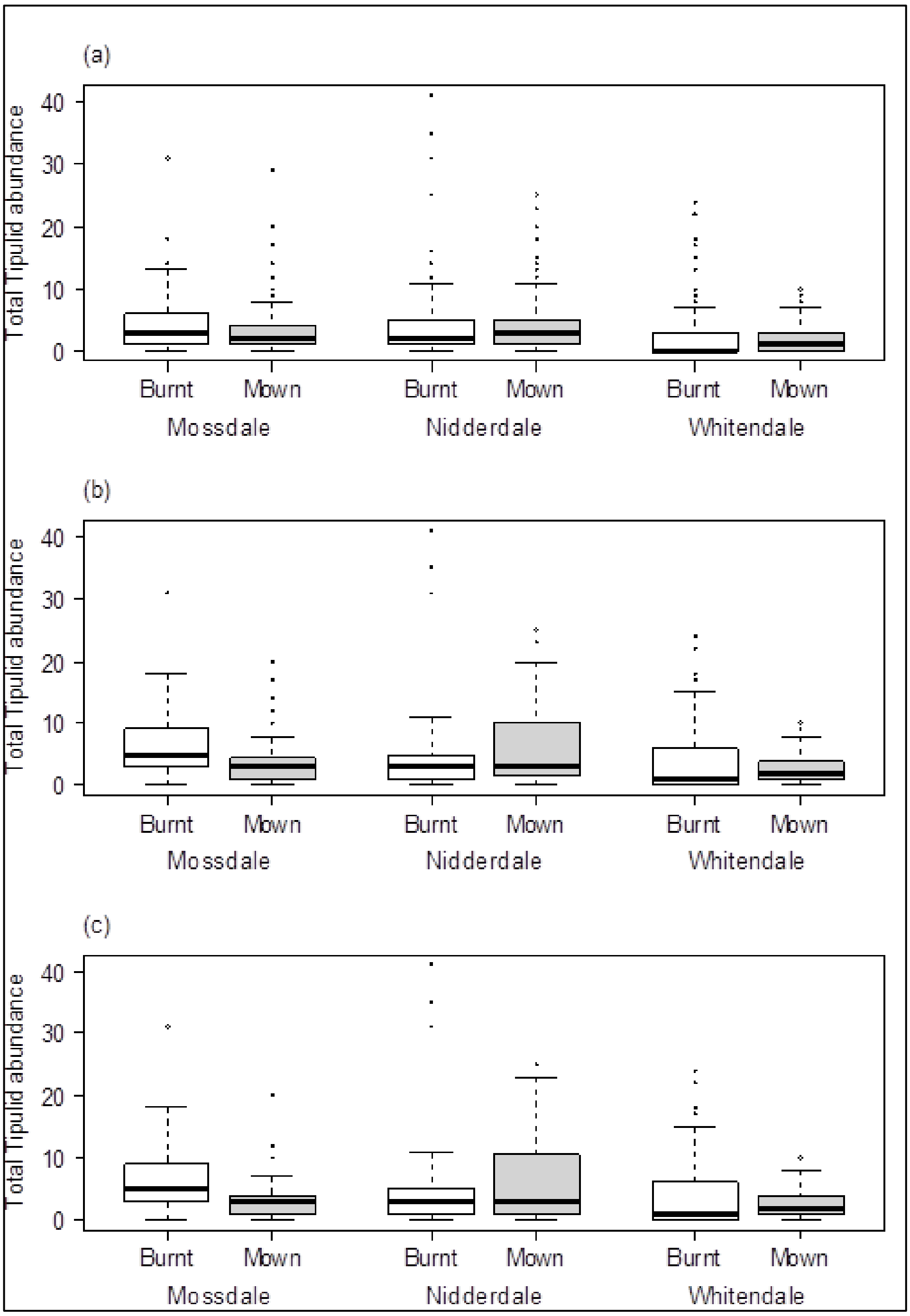


Fig. 119 Comparison of total cranefly (Tipulid) trap emergence (abundance) between burnt and mown sub-catchments for separate sites, using (**a**) all data; (**b**) data just from plots (not slopes); and (**c**) also excluding traps in uncut (DN) plots. Box midline indicates median, box edges indicate interquartile range, whiskers indicate range of data and points indicate data outside 1.5 x the interquartile range.

4.5.4 Effect of plot-level management (uncut, brash removal) on soil moisture and cranefly trap emergence

Whereas the previous analysis (Sections 4.5.2 and 4.5.3) focused on the burnt and mown sub-catchments including slope locations, a more detailed plot-level management cranefly trap emergence analysis was also performed. This specifically considered the uncut and the two mowing with or without brash removal treatments within the mown sub-catchment.

Although median soil moisture was highest for mown plots with brash left across all years and site, there was no significant difference between either soil moisture or cranefly trap abundance when comparing the uncut plots with the two mowing plot level managements (cut with brash removed and cut with brash left) across all sites and years (**Figure 120**).

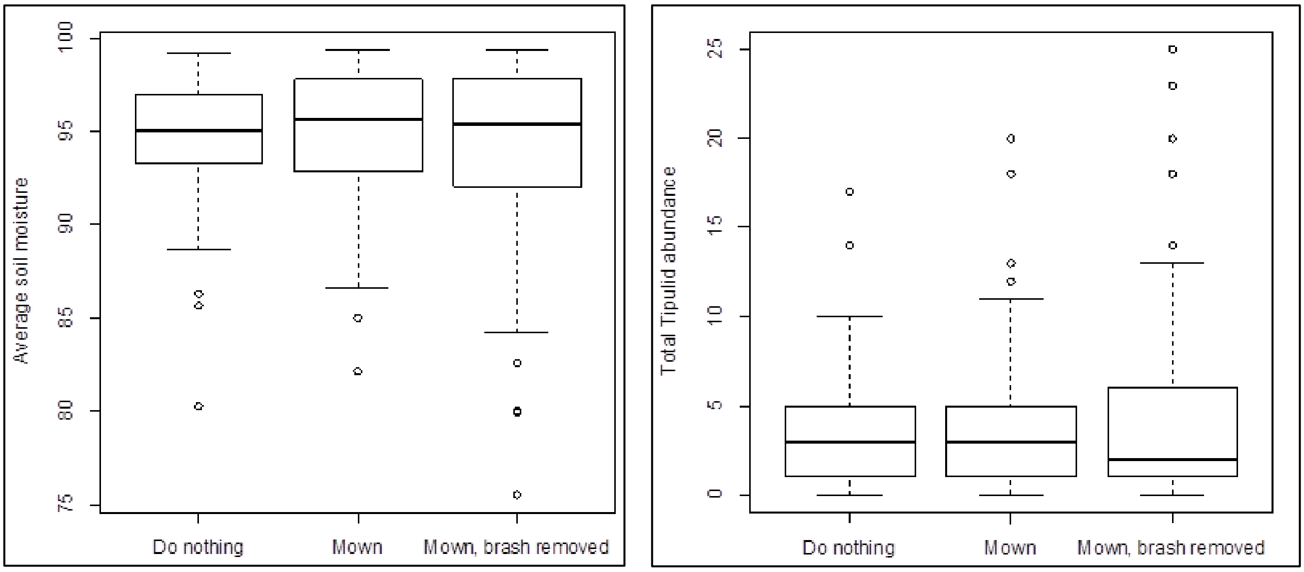


Fig. 120 Annual mean soil moisture (%; **left**) and total cranefly (Tipulid) trap emergence counts (abundance) (**right**) in plots where the management was: uncut (do nothing), mowing with brash left (LB) or mowing with brash removed (BR). Box midline indicates median, box edges indicate interquartile range, whiskers indicate range of data and points indicate data outside 1.5 x the interquartile range.

Notwithstanding the complex nature of overall moisture impacts on cranefly emergence, the observed cranefly trap emergence data allow predictions to be made of average cranefly emergence for different managements (**Figure 121**) based on statistical models (see Appendix 11 for a more detailed description). The statistical model identified significant interactions between year ($p< 0.001$) and both period and treatment ($p< 0.001$), as well as impacts of plot versus slope ($p< 0.001$) and air temperature ($p< 0.001$) upon cranefly abundance. Back transformation of the parameter estimates from this model yielded an average 0.36 craneflies per trap within the burnt (95% confidence interval (CI) 0.31 - 0.43) and 0.32 (95% CI 0.28 - 0.38) per trap in the mown catchment. Although the analysis indicated that management did not have an overall impact on the abundance of emergent craneflies ($p = 0.568$), plotting the annual variation in estimated effects suggested that there was a significant positive effect of mowing treatment upon cranefly emergence in the first year (2014) by 50%, no significant impact in year 2, and a negative effect of treatment in the third year (2016) by 50% (**Figure 121**; left). The reason to also consider slope and uncut plots in the analysis reflects that plots within each sub-catchment are hydrologically connected to the overall burnt or mown catchment. Notably, the uncut plots fall in between burnt and mown ones, which most likely relates to some mowing impact on the uncut plots (i.e. they are surrounded by wetter mown areas). Moreover, to capture the overall management related impact on cranefly emergence, the to adult (i.e. flying) craneflies available entire catchment area not just individually managed plots within a catchment should be considered.

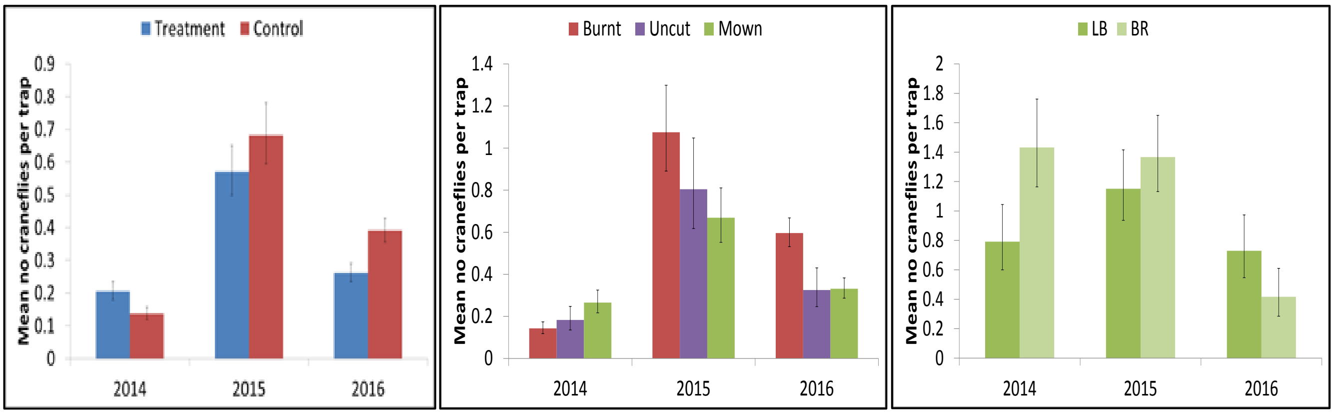


Fig. 121 Modelled effect of (**left**) overall mowing (treatment) versus burning (control) sub-catchments, burnt versus uncut (plot only) versus mown (combined with or without brash removal) management (**middle**), and (for the combined ±*Sphagnum* additions) left brash (LB) versus brash removal (BR) (**right**) upon the (back transformed) abundance of craneflies in emergence traps in each of the three years of the study. Estimated effect sizes are from a model that accounts for temporal variation in phenology between years (apart from for brash removal – due to model instability). Error bars show SE.

This analysis was refined further by separating data from individual traps into those from burnt, mown and uncut areas, which identified significant interactions between year and management ($p < 0.001$), and year and period ($p < 0.001$). This re-enforced the observed management effect from across the sub-catchments, where in the first (driest) year, mown areas were associated with the highest emergence of craneflies and newly burnt areas, the lowest (**Figure 121**; middle). However, this pattern was reversed in the wetter years 2015, when counts were higher and more variable, and particularly in 2016 when the greatest numbers of craneflies were recorded from the burnt plots (see Appendix 11 for more detailed analysis output). In order to test the potential for brash removal to alter cranefly abundance, an additional analysis of variation in cranefly emergence in traps within mown treatment areas was conducted. The model output indicated a significant interaction between brash removal and year ($p = 0.011$), such that cranefly emergence was higher from burnt plots with brash removal in 2014, but lower by 2016 (**Figure 121**; right). One plausible cause could be that mowing with brash removal opens the sward, enabling easier oviposition by females into the peat (McCracken & Tallowin 2004). The immediate negative effect of burning may reflect a fire impact on survival of craneflies, either directly or indirectly through impacts on soil moisture (Coulson, 1962).

In summary, although soil moisture was consistently higher under mowing than burning, the pattern for cranefly trap emergence was more complex. One explanation for this is that mowing buffered peat against drying out in dry years with potential positive impacts for craneflies, but potentially made it too wet in wet years, with negative impacts on cranefly larvae and consequently emergence. Unfortunately, although future climate predictions point towards drier summers, only one dry year (2014) was captured in this study and immediately after management. However, surprisingly little is known about water table and moisture effects on actual survival rates of cranefly larvae. So far, only one study is available which assessed survival in the field in relation to desiccation (Coulson, 1962), which is surprising considering the well-known importance of craneflies for upland birds (Carroll et al., 2011). Clearly more experimental and inter-annual monitoring data on impacts of peat water table and moisture on cranefly larvae survival and emergence are needed. However, the analysis of soil moisture and cranefly emergence traps highlighted the importance of soil moisture and interactions with management, which was explored further in two model approaches. One approach was based on cranefly abundance transect of catchment-scale management (i.e. burning versus mowing), another approach combined trap emergence data with peat model predictions of water table and soil moisture, which considered plot-level management impacts at the landscape scale within future climate scenarios.

4.5.5 Predicting impacts on birds

Craneflies are important food for upland birds (Buchanan et al., 2006; Pearce-Higgins, 2010), and particularly in their chick phase (Park et al., 2001; Pearce-Higgins & Yalden, 2004). Importantly, direct moisture and moisture affecting management impacts on craneflies, as shown by Carroll et al. (2011), are key in predicting impacts of both management and climate on bird populations. The observed management and environmental impacts on cranefly trap emergence and transect abundance were combined with established links to upland bird species. This allowed making model predictions about the implications of both, management and climate, using two approaches. The first two cranefly approaches focused on the two main managements (using transect data across burnt and mown sub-catchments) and impacts on just golden plover, the second cranefly emergence approach considered individual plot-level management impacts (using traps) on moisture and craneflies together with model predicted changes in water tables reflecting future climate (drier summers) scenarios and predicted the impacts on three upland bird species (i.e. golden plover, dunlin and red grouse).

4.5.5.1 Cranefly transect approaches: predicting impacts on golden plover

Model of golden plover productivity and population change

The abundance of adult (sitting and/or flying) craneflies was measured along transects (10 m long, 4 m wide) across the entire sub-catchments. In order to assess the effect of management upon the cranefly abundance observed on transects cranefly count was modelled as a function of the main catchment-scale management only (mown versus burnt sub-catchment), considering plot type (plot versus slope), year (2014, 2015, 2016), survey period (four in 2014 and 2016, five in 2015) and air temperature, the latter to account for variation due to ambient conditions. The model identified significant effects of management ($p< 0.05$), plot ($p< 0.01$), year*period ($p< 0.001$), and temperature ($p< 0.001$), but with no significant effect of year on the management or plot effects. Back transformation of the parameter estimates yielded an average 0.55 craneflies per burnt (95% CI 0.43 - 0.70) and 0.81 (0.64 - 1.03) per mown sub-catchments per 10 m transect. The key conclusion of this analysis was that 67% more craneflies were recorded from transects on mown compared to burnt catchments, with no significant variation between years (Appendix 11 provides further detailed methods and statistical output).

These estimates of cranefly abundance along 10 m transects are broadly comparable with those from 20 m transects used by Pearce-Higgins & Yalden (2004) to link cranefly abundance to golden plover chick survival. In order to link variation in cranefly abundance to golden plover productivity, as an index of wider ecological impacts, it is necessary to model the daily encounter rate of golden plover chicks with craneflies throughout the season. This was done by using the year and period interaction to model temporal variation in cranefly abundance in each year, and then incorporating the additive impact of management onto that, assuming (based in the analysis described above) that phenology varies between years, but not between treatment (mown) and control (burnt) catchments. This model contained a significant effect of treatment ($F_{1, 71.03} = 11.72$, $p = 0.001$) and plot ($F_{1, 69.46} = 9.49$, $p = 0.003$), and a non-significant effect of year*period ($F_{12, 1} = 49.56$, $p = 0.11$). The resulting predictions were used to create a daily cranefly profile per year per management (**Figure 122**), which showed a similar overall management effect as that described above from the overall model with temperature; 0.55 craneflies per 10 m transect on burnt (95% CI 0.47 - 0.75) and 0.91 (0.73 - 1.15) on mown sub-catchments.

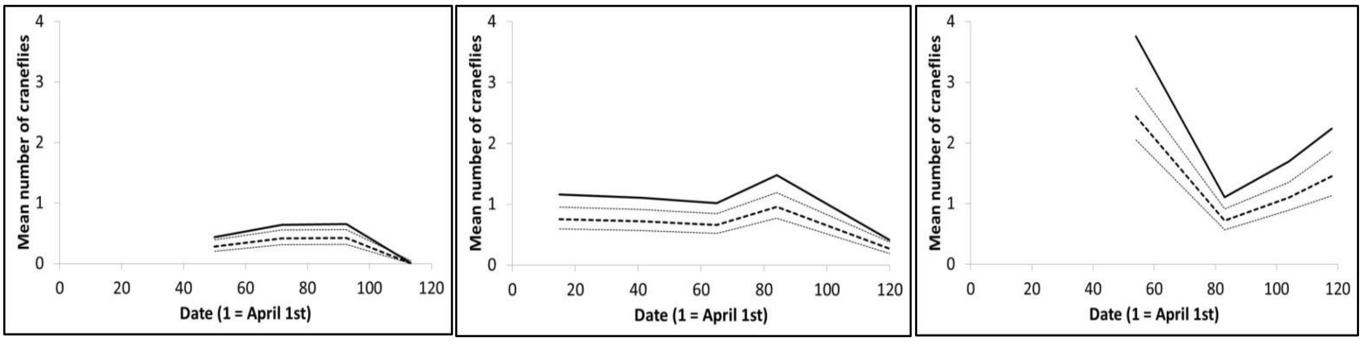


Fig. 122 Modelled temporal variation in cranefly abundance (per 10 m transect) in 2014 (**left**), 2015 (**middle**) and 2016 (**right**) in control burnt (dotted-line) and mown (solid-line) sub-catchments. Estimates for the control are calculated from a model that tests for significant differences between mown and burnt sites, and hence 95% CI (thin lines) are only produced for the burnt data. When the treatment line falls outside those 95% CI, there is a significant effect of management.

The transect data suggested a stronger impact of mowing upon cranefly abundance of around 50% higher abundance when averaged across the entire dataset (**Figure 122**). This difference from the trap data may result from the fact that the abundance of craneflies along transects is not just a function of local emergence, but also the movement of emerged craneflies. Mowing may therefore produce a micro-habitat that is more conducive to the aggregation of adult craneflies than burning, perhaps due to the residual vegetation structure which may provide shelter for individuals, or provide a more suitable microclimate than burnt areas. Although less than the 2-5 times increase in cranefly abundance associated with blocking drainage ditches compared to open ditches (Carroll et al., 2011), the magnitude of this increase does suggest that burning can have a negative impact on cranefly abundance locally, at least in the first year following burning, and that in the longer term, mowing can increase the abundance of craneflies for foraging birds. It is likely that the mechanism for the immediate response may be a function of microclimate, with burning likely to be associated with reduced soil moisture compared to cut plots, but the most likely hypothesis for the longer-term effect is that mown plots attract or retain adult craneflies more than burnt plots. These two mechanisms could account for the different management effects observed between the traps and transects. Importantly, as so far less than 50% of the entire catchment area has been managed differently, monitoring over a full management cycle might reveal a much stronger relationship.

The golden plover model predicts daily survival of young (<9 day old, when most mortality occurs; Pearce-Higgins & Yalden 2004) golden plover chicks as a function of daily cranefly abundance. The daily predictions of cranefly abundance for mown and burnt sub-catchments in each year described above were used to predict daily golden plover chick survival, assuming a constant golden plover nesting phenology matching that from Pearce-Higgins (2011). Variation in daily cranefly abundance is a function both of the phenology of cranefly emergence and overall cranefly abundance. Both vary widely between years, for example as a result of variation in temperature (Pearce-Higgins et al., 2010), as shown by **Figure 122**. These predicted values of daily cranefly abundance (**Figure 122**) were used to model the impact of management upon golden plover breeding success, after accounting for the background variation in cranefly abundance and phenology.

Significantly higher golden plover productivity (fledglings per 100 pairs) on mown than burnt management were predicted in all years (**Figure 123**, left), reflecting the differences in cranefly abundance. These differences were greatest in 2014, when overall cranefly abundance was lowest, and were too small to be biologically meaningful in 2016, when overall cranefly abundance was highest (**Figure 122**). The differences between years were consistent with the hypothesis that the effect of management upon cranefly abundance may be increasingly important in years of low overall abundance (when it is the main driver of bird productivity) rather than high overall abundance (when food is not limiting; Pearce-Higgins, 2011). The same patterns are reflected in predictions of golden plover population change with fledging production above replacement in all years and treatments (**Figure 123**, right). The observed increased cranefly and golden plover abundance due to mowing

compared to burning in dry years is therefore an important management tool when considering how to increase the resilience of blanket bog functioning under climate change scenarios.

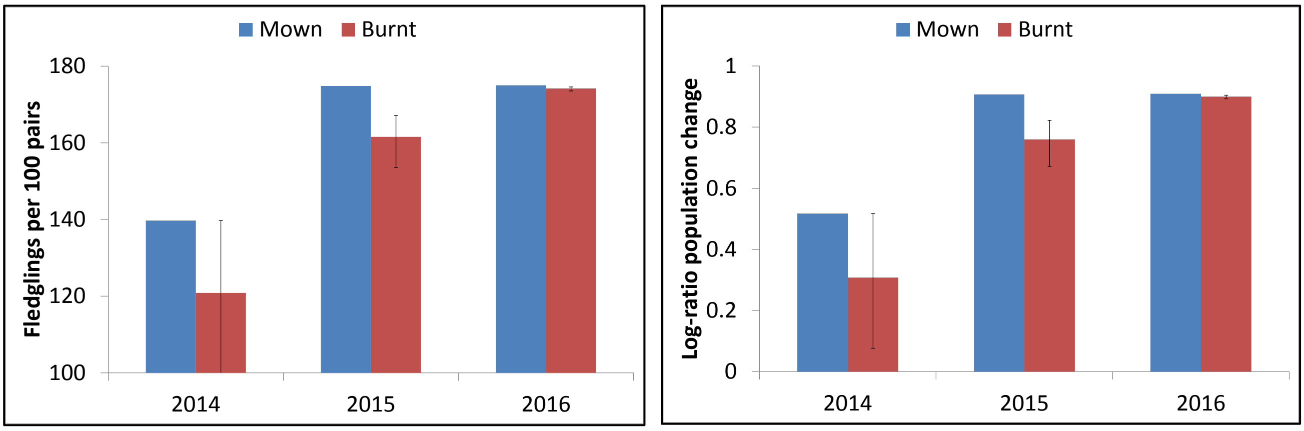
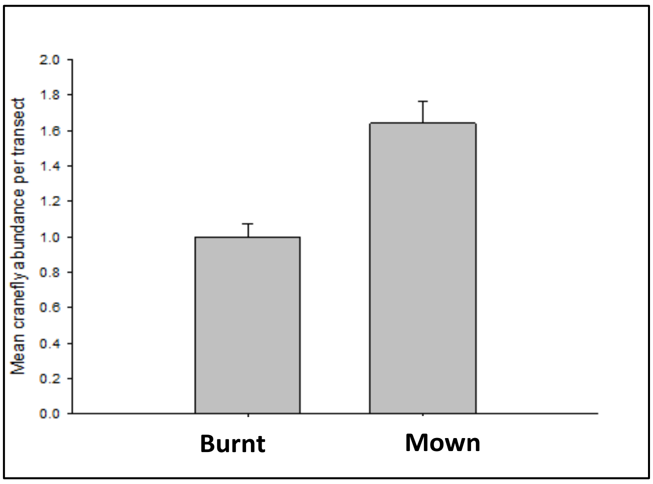


Fig. 123 Modelled variation in golden plover productivity (**left**) and in golden plover population change (ln (ny+1/ny)), where population change is the count (n) in year (y+1) over the count in year y (**right**) based on modelled cranefly abundances in plots within the treatment (mown) and control (burnt) catchments, with annually varying background cranefly abundances and phenologies. The error bars for the control plots indicate 95% confidence interval relative to the treatment estimates, and demonstrate significant differences in all cases.

The model underpinning **Figure 123** also assumed that estimates of nest and chick predation match those from the Pearce-Higgins (2011) study; as both the location of that study, and these experimental plots were from grouse moors, the assumption was not unrealistic. Moreover, predictions were based upon 100% burnt or mown habitat, and did not take account of patchiness and changes in vegetation height. The modelling suggested that an increase in cranefly abundance led to a statistically significant increase in golden plover abundance, although the magnitude of this increase was most apparent in 2014, which was the year of lowest cranefly abundance, translating to an up to 8% increase in breeding success, and a subsequent impact on predicted population growth rates. It is worth noting that cranefly abundance in each of these three years, but particularly 2015 and 2016, was high compared to previous studies (Pearce-Higgins & Yalden, 2004; Pearce-Higgins, 2011). This is important because, when cranefly abundance is high, variation in cranefly abundance is not limiting for golden plover. As a result, our analysis likely underestimated the impact of any benefit of increased cranefly abundance on golden plovers compared to other sites with lower availability of craneflies, or in a drier site or year context.

Model of golden plover fledging young

The cranefly data were also used to develop an overall model of golden plover fledging young and of breeding pair density including vegetation height effects. Firstly, the overall transect cranefly abundance was first compared between burnt and mown sub-catchments. The final model revealed a significant catchment-scale management effect on cranefly abundance ($\chi^2 1 = 9.40$, p = 0.002), with no evidence of significant difference between years (treatment x year $\chi^2 1 = 0.97$, p = 0.614) but an overall air temperature effect ($\chi^2 1 = 84.58$, p< 0.001). Mean fitted counts per transect were: burning = 1.00, 95% CI = 0.92 – 1.07; mowing = 1.64, 95% CI = 1.51 – 1.77 (**Figure 124**, left). Thus mean cranefly abundance is estimated to be 64% higher within mown treatments than burning (similar to **Figure 122**). Secondly, golden plover productivity (the probability of a pair fledging young in each treatment) was predicted using the management-specific values of cranefly abundance. This approach used a Minimum Adequate Model produced by Douglas & Pearce-Higgins (2014), which relates the probability of a pair of golden successfully fledging young to cranefly (transect) abundance, using appropriate transformation for predicting probabilities bounded between 0 and 1 (**Figure 124**, right). The probability of a pair fledging young within the burnt management was predicted to be 0.55 (95% CI = 0.53-0.58), and within the mowing treatment as 0.73 (0.70-0.76). Thus fledging probability per pair is predicted to be 33% higher in the mown compared to the burnt management based on higher modelled catchment-scale cranefly abundance.



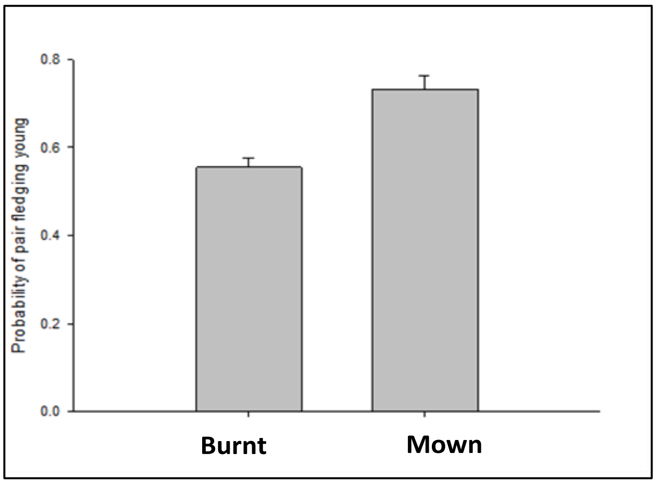
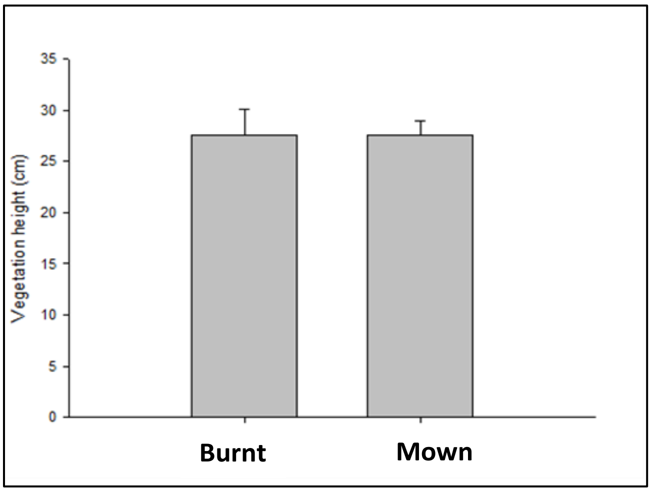


Fig. 124 Mean + 95% confidence interval for (**left**) cranefly abundance per transect within the burnt or mown sub-catchments, and (**right**) GLMM predicted probability of a pair of golden plover successfully fledging young, based on management-specific values of cranefly abundance based on transect abundance surveys.

Model of number of breeding pairs

Finally, an analysis of variation in vegetation height between burnt and mown sub-catchments allowed using management-specific values of vegetation height to predict golden plover breeding pair density per treatment. For this modelling exercise, vegetation height measurements (i.e. not just heather height) were made in monitoring plots in 2016 (see Appendix 11 for further details). The maximum height of any vegetation was recorded in 2016 for 20 random positions across each 5 x 5 m plot and along the 10 m transects. Comparable average values per plot for 2014 and 2015 were based on best estimates of expected vegetation height (using photographs to estimate an overall growth increase over time; any negative values were replaced with zero in the analyses). The model showed that vegetation height did not differ significantly between catchment management ($F_{1, 233}$ = 0.64, p = 0.424) (**Figure 125**; left). Subsequently, a Minimum Adequate Model (Douglas & Pearce-Higgins, 2014) related the breeding pair density of golden plover to vegetation height (controlling for altitude). Parameter estimates from this model were combined with management-specific means of vegetation height from the current study, assuming a standard 420 m altitude across treatments (median of 390 – 450 m range). Predicted breeding pair densities were back-transformed using an exponential transformation to ensure predicted count values were non-negative. Predicted breeding densities were identical: 0.33 pairs km^{-2} (95% CI = 0.20 - 0.53) within the burnt and 0.33 (0.25 - 0.43) within the mown management catchments (**Figure 125**; right).



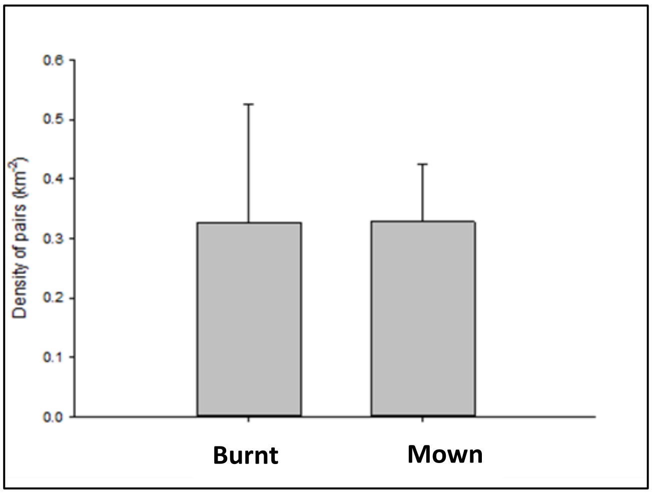


Fig. 125 Mean + 95% confidence interval for (**left**) vegetation height and (**right**) corresponding predicted golden plover breeding pair density between sub-catchment management of burning versus mowing predicted from GLM based on modelled vegetation height based on relationship between golden plover breeding pair density and mean vegetation height within 1 km^2 survey squares as in Douglas & Pearce-Higgins (2014).

It should be noted that modelled sub-catchment vegetation heights in this study were relatively high and predicted breeding pair densities relatively low when compared to the study of Douglas & Pearce-Higgins (2014), and predictions were thus extrapolated just beyond the range of vegetation height values from Douglas & Pearce-Higgins (2014).

A final analysis was made of differences in vegetation height between different plot treatments, rather than considering the catchment-scale mowing and burning comparison. The analysis revealed a highly significant treatment*year interaction in vegetation height ($p < 0.001$), related to the vegetation regrowth on both burnt and mown plots in comparison to the uncut plots (**Figure 126**; left). The management and year-specific mean vegetation heights were then used to predict golden plover breeding pair densities over time using parameter estimates from Douglas & Pearce-Higgins (2014). This showed that potential golden plover densities are predicted to be relatively high in the early post-management period on both mown and burnt plots, where vegetation was much shorter than in the uncut plots, declining to similarly low values as for the uncut plots by 2016 as vegetation height increased during the post-management period (**Figure 126**; right).

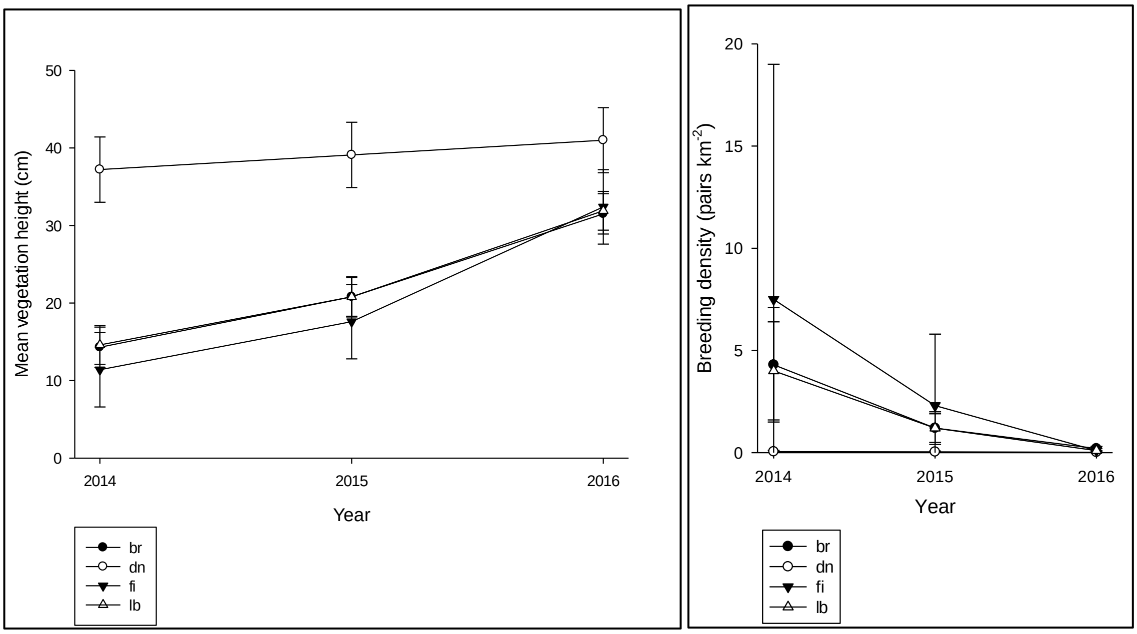


Fig. 126 Change over time in (**left**) vegetation height and (**right**) breeding pair density of golden plover under different managements: mown with brash removed (br), uncut (dn), burnt (fi) and mown with left brash (lb) treatments predicted from GLMM showing mean values ± 95% confidence interval.

In conclusion, cranefly abundance modelled using transect counts suggested that mean abundance across the three years of the study was 64% higher in the mown compared to the burnt sub-catchments. This translated into the predicted probability of a pair of golden plover successfully fledging young to be overall 33% higher in the mown treatment over the three-year period. This analysis has tested, for the first time, the potential impact of mowing and burning management upon cranefly abundance, and considered the potential impacts on breeding birds. Modelled management differences in vegetation height were very small (and statistically non-significant) when considering a burnt and mown catchment-scale comparison. Coarse examination of vegetation heights between the plot-level mown sub-catchment managements suggested a variation in vegetation height that may merit further examination, and could yield more detailed assessment of potential variation in golden plover breeding density. In particular, considering a realistic patchwork of vegetation heights (i.e. reflecting recently or regrown burnt, mown and mature patches) across the landscape seems important to provide practitioner relevant scenario information. Moreover, impacts on other breeding birds, including red grouse, have not been modelled to date due to a lack of specific model parameter data. A general question remains as to how much tall (heather) vegetation is needed by red grouse and other upland bird species to provide food as well as shelter.

4.5.5.2 Cranefly emergence trap approach: predicting impacts on golden plover, dunlin and red grouse

Predicting landscape scale cranefly abundance

Cranefly larvae are affected by moisture changes throughout their life cycle (Coulson, 1962), particularly during the summer (dry) and winter periods (wet) before their emergence the following year. This study could only compare soil moisture data which were paired with cranefly emergence data observed at the same time. Ideally, cranefly data would be related to moisture data during the previous year. However, peat model predictions of water table depth (WTD) are a suitable tool to address this issue across space and time (Carroll et al., 2015) by incorporating empirical relationships between soil moisture and water tables. In order to predict cranefly numbers across our three focal landscapes (5 x 5 km squares containing our three field sites) three steps were performed, we:

(1) modelled water table depth across space and time for a baseline and future using the MILLENNIA peat development model (see Carroll et al., 2015);

(2) predicted soil moisture based on an empirical relationship between soil moisture (%) versus water table depth (cm) established using field data; and

(3) predicted cranefly abundance across the landscapes based on the relationship between cranefly emergence and soil moisture we established from our field trap data.

Peat model water table validation

In addition to the previous model validation (Carroll et al., 2015), the three sites offered an independent validation based on plot-level WTD data. Since the previous publications (Heinemeyer et al., 2010; Carroll et al., 2015), the MILLENNIA model now also includes a bedrock drainage factor, derived from information on specific yield and hydraulic conductivity (see Appendix 15). This allowed the model to better reflect overall observed site differences, most likely reflecting drainage through the peat/bedrock interface, which also includes underground drainage via peat pipes. Although peat pipe numbers were similar between sites (**Table 4**), the bedrock layer visually differed in porosity (grain size), and this could have been reflected in the slightly lower WTD at Nidderdale and Whitendale observed compared to model predictions (see **Figure A15.2** in Appendix 15). However, other factors such as site-specific evapotranspiration in relation to vegetation and sunshine hours, and cloud or fog cover, could have affected the observed hydrological differences. Therefore, slight adjustments to the default parameters of the drainage parameters, hydraulic conductivity and specific yield, were made to obtain a better model fit (**Figure 127** cf. **Figure A15.2** in Appendix 15).

Overall, both model predictions covered the seasonal changes very well. However, the adjusted model (**Figure 127**) improved the match to observed WTD although it caused an overall lower predicted WTD for Nidderdale and Whitendale but only in 2016. The most likely reason is that the monthly model predicted too high a monthly runoff, as it did not capture the effective infiltration rates of daily rainfall (which was consistently high in the summer 2016; with monthly rainfall totals shown in **Figure 10**). High monthly totals of rainfall and high overall WTD predict higher runoff in the model, leading to lower overall WTD. Although similarly high rainfall occurred during 2012, peat WTD was overall lower at the start of the year (2011/12), thus allowing greater infiltration avoiding over-prediction of runoff. The improved and adjusted model therefore captured the overall WTD pattern better and was applied within this study.

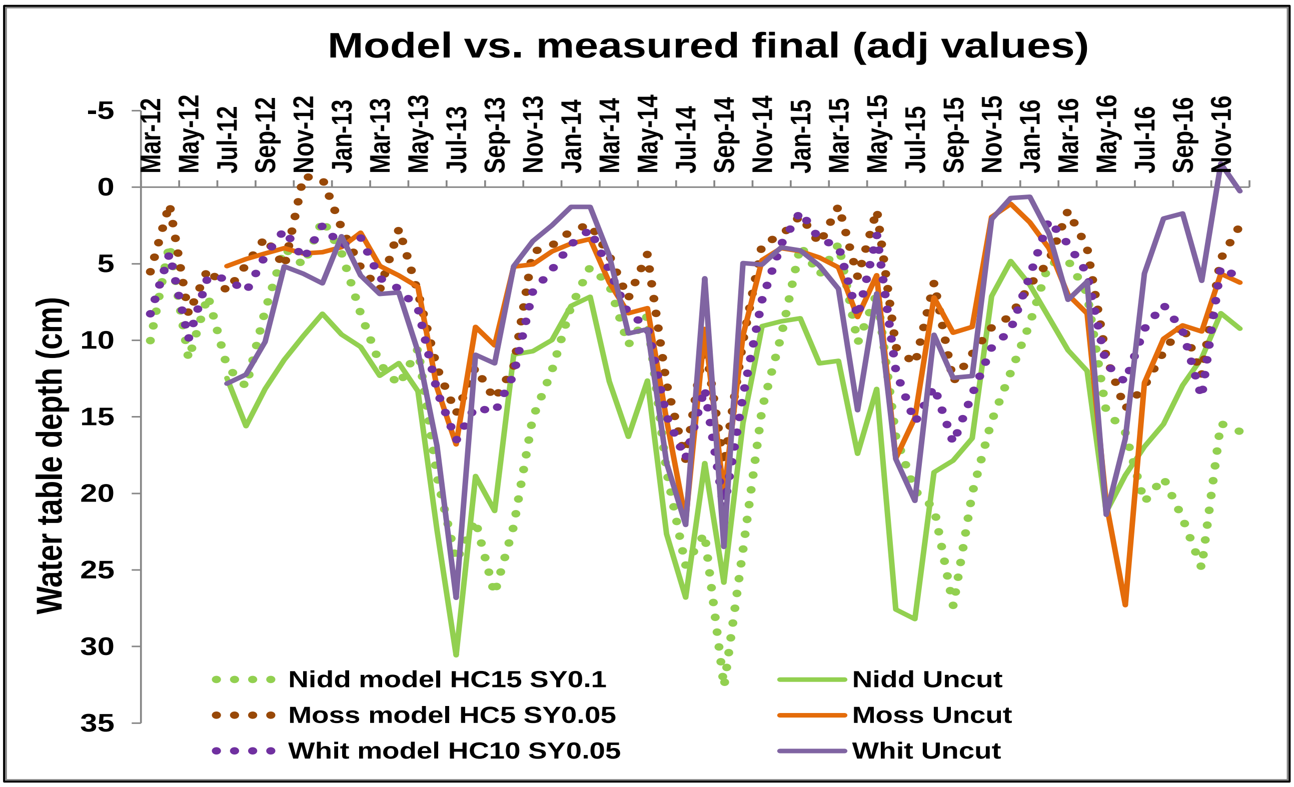


Fig. 127 Site averages versus model predicted mean monthly water table depths for the three sites (Nidderdale, Nidd; Mossdale, Moss; Whitendale, Whit) for the uncut management with the adjusted parameters (adj. values) for hydraulic conductivity (HC) and specific yield (SY). The default values (i.e. as for Mossdale) were HC = 5; SY = 0.05.

Modelled peat hydrological site conditions

The adjusted MILLENNIA model was used to enable plot-level observed WTD to be up-scaled across catchments to predict hydrological impacts on cranefly and subsequently bird populations. Similar work was done previously (Carroll et al., 2015) showing very good agreement between modelled and measured WTD at three UK sites. In particular, the model is also able to capture impacts of topography (i.e. slope) in relation to runoff and increased evapotranspiration (i.e. aspect). For all three sites, WTD values were predicted across the wider 5 x 5 km catchment-scale (**Figure 128**). This showed that the paired sub-catchments covered very similar ranges of mean WTD and that there was an overall lowering in WTD (i.e. sites became drier) from Mossdale to Whitendale to Nidderdale (**Figure A15.1** in Appendix 15 provides an example for a 50 m grid resolution for 2014). However, whereas predicted peat areas below an arbitrary 200 m elevation limit were excluded (as these reflected grass meadows with sheep grazing on no peat soils) the model also predicted WTD for peat on steeper slopes with WTDs at the bedrock (see red colours in **Figure 128**). These areas were included in the predictions as they reflect naturally dry slopes with peaty soils and are likely to apply across the catchment under a future drier climate.

The model was then used to predict long-term WTDs of baseline climate (1961-1990) and projected future climate change (2051-2080) across each catchment using the same method as outlined in Carroll et al. (2015). Further model and scenario details are also provided in Appendices 10 and 15. The baseline climate captured the observed site differences, with Mossdale being the wettest site, Whitendale intermediate and Nidderdale being the driest site (**Figure 128**). This reflected the overall observed site differences in climatic conditions during the 5-year study, particularly decreasing rainfall amounts (2000 mm, 1850 mm and 1600 mm respectively; **Table 1**).

The future climate change scenarios predict a pattern of increasing rainfall over winter months with decreasing amounts in summer. Future projections of mean annual WTDs reflect this, showing an overall drier habitat, particularly on slopes and south facing areas (**Figure 128**). However, the overall effect of this lowering in WTD differed between sites. Whereas Mossdale showed very little change in WTD overall, Nidderdale and Whitendale both showed a large lowering in WTDs (i.e. drier) across the catchments. Whereas annual mean WTDs at

Mossdale remained within the 10 - 20 cm range in upland areas, WTD values at both Nidderdale and Whitendale fell to less than 20 - 30 cm in most upland areas and well beyond this in lower areas and regions of steeper slopes. Clearly, the blanket bog habitat is predicted to shrink under this future climate scenario, particularly for the sites below a current average rainfall of 2000 mm. This is similar to climate change impacts on UK blanket bogs that were predicted by Clark et al. (2010) based on climate envelope models.

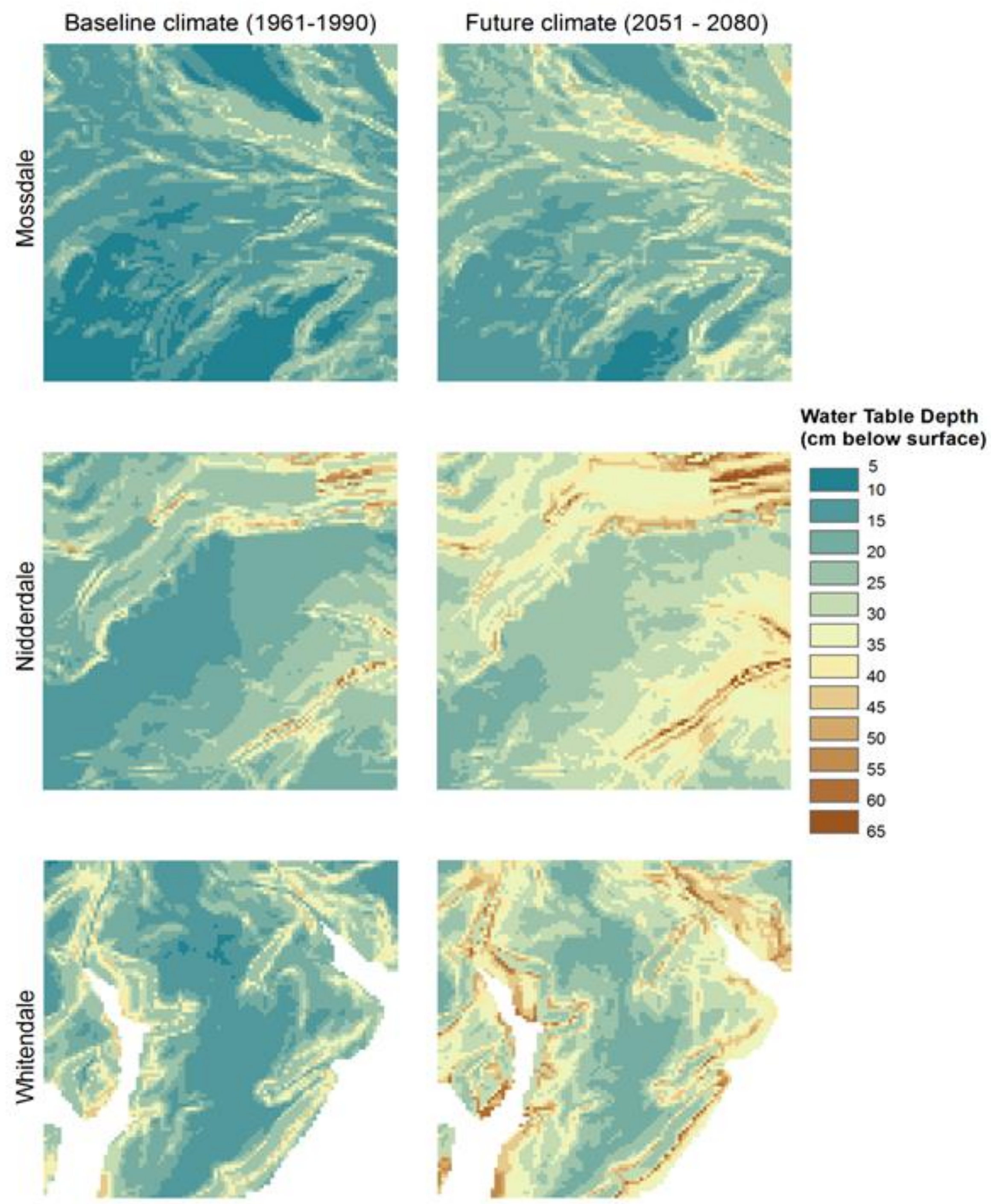


Fig. 128 Annual mean water table depth (WTD) predictions for the 5x5 km squares for each site for (**left**) the current baseline (1961-1990) and (**right**) future climate (2051-2080) for the cranefly scenarios as shown in the **Figures 131-133**. Whitendale areas under 200 m a.s.l. were not included in the model as they do not support observable blanket bog peatland habitat.

Water table depth (WTD) predictions (step 1) were obtained for each 50 x 50 m square (considering aspect, slope and elevation effects on peat hydrology) from the adapted monthly version (Carroll et al., 2015) of the MILLENNIA peatland model (Heinemeyer et al., 2010). Specifically, summer WTD (July-September) was calculated, reflecting the observed cranefly desiccation period (Coulson, 1962). Two scenarios were considered: baseline climate (1961-1990) and a future (2051-2080) climate scenario (SRES emissions scenario A1B; UKCP09). Further adjustments to the mean summer WTD predictions were made for management using the annual mean WTD offsets of different managements relative to the uncut 'do nothing' management based on the observed data (**Table 9**). However, the more conservative (i.e. less dry) burn impact of a reduction in WTD of only 1.7 cm was used (i.e. excluding one plot at Whitendale with overall very low WTD).

For the relationship between soil moisture and water table depth (WTD) (step 2), soil moisture readings taken at the cranefly trap locations were compared with the automated dipwell readings taken from the corresponding plots for the same day. Overall, there was a clear negative polynomial relationship, with a sharp decrease in soil moisture when water tables were below 10-20 cm (see **Figure 129**). However, rainfall shortly before soil moisture measurements was found to increase soil moisture disproportionately compared to largely unchanged water tables (i.e. requiring more substantial rainfall). Therefore, the model fit was improved by excluding data with significant (>10 mm) rainfall over the previous three days which was applied to all models. All models were also set with an intercept of 1.0, as soil moisture will be 100% when the water table is at the surface (0 cm).

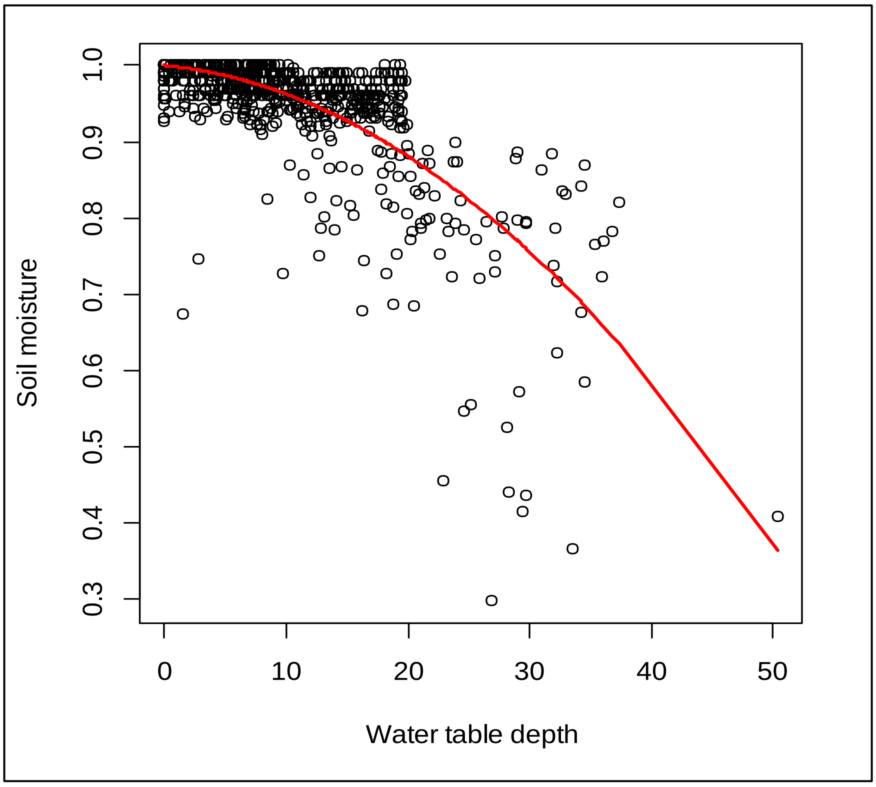


Fig. 129 Relationship between soil moisture (%) and water table depth (cm) across all sites and combined treatments and years. Days with significant amounts of rainfall over the previous 3 days were removed. Regression equation: soil moisture = 1.00 – 0.00143(WTD) – 0.000239(WTD^2); R^2 = 0.449, P< 0.001.

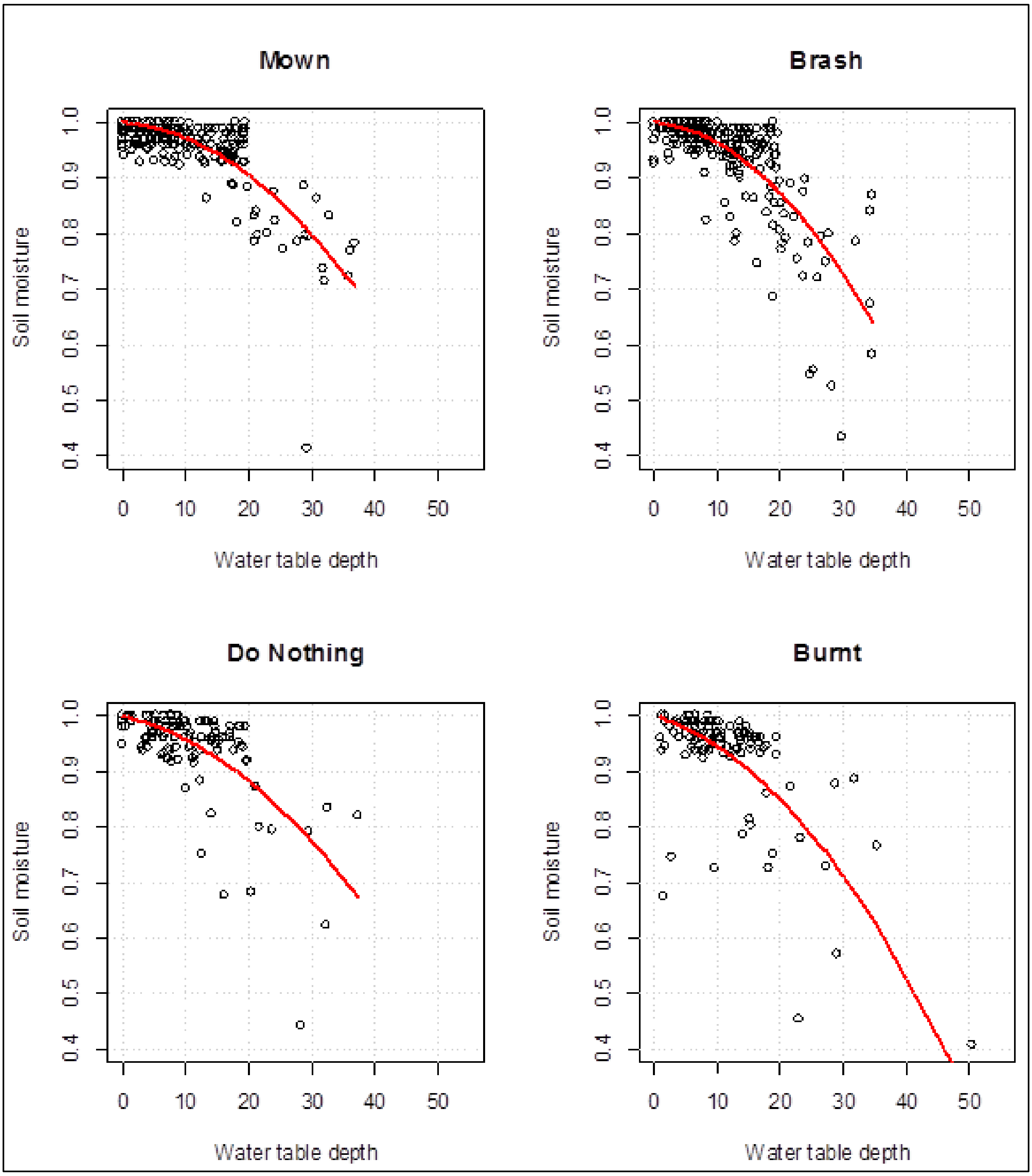


Fig. 130 Relationships between soil moisture (SM, %) and water table depth (cm) across all sites, and years for each (plot) management. Days with significant amounts of rainfall over the previous 3 days were removed. Regression equations were: Mown (brash left): SM = 1.00 – 0.000758(WTD) – 0.000197(WTD^2), R^2 = 0.557, P< 0.001; mown brash removed (Brash): SM = 1.00 – 0.000903(WTD) – 0.000275(WTD^2), R^2 = 0.550, P< 0.001; uncut (Do Nothing): SM = 1.00 – 0.00256(WTD) – 0.000165(WTD^2), R^2 = 0.411, P< 0.001; Burnt: SM = 1.00 – 0.00342(WTD) – 0.000205(WTD^2), R^2 = 0.400, P< 0.001.

A comparison of the relationship between soil moisture and WTD across all sites revealed clear differences between management with a highly significant (p< 0.001) and overall good model fit (mean r^2 of 0.5) for all managements (**Figure 130** above). The steepest decline was observed for brash removal and burnt managements.

Although the effects of management were consistent, the best model fits for each management varied between sites as follows (see Appendix 10 for all analysis output):

Nidderdale:
Mown (LB): $SM = 1.00 - 0.00379(WTD) - 0.0000487(WTD^2)$; $R^2 = 0.794$, $p < 0.001$
-Brash (BR): $SM = 1.00 - 0.00374(WTD) - 0.000185(WTD^2)$; $R^2 = 0.495$, $p < 0.001$
Uncut (DN): $SM = 1.00 - 0.000494(WTD) - 0.0000835(WTD^2)$; $R^2 = 0.603$, $p < 0.001$
Burnt (FI): $SM = 1.00 - 0.00329(WTD) - 0.000171(WTD^2)$; $R^2 = 0.647$, $p < 0.001$

Mossdale:
Mown (LB): $SM = 1.00 + 0.000342(WTD) - 0.000263(WTD^2)$; $R^2 = 0.576$, $p < 0.001$
-Brash (BR): $SM = 1.00 - 0.00161(WTD) - 0.000191(WTD^2)$; $R^2 = 0.594$, $p < 0.001$
Uncut (DN): $SM = 1.00 + 0.0012(WTD) - 0.000424(WTD^2)$; $R^2 = 0.494$, $p < 0.001$
Burnt (FI): $SM = 1.00 - 0.00655(WTD) - 0.000136(WTD^2)$; $R^2 = 0.421$, $p < 0.001$

Whitendale:
Mown (LB): $SM = 1.00 + 0.00176(WTD) - 0.0000315(WTD^2)$; $R^2 = 0.732$, $p < 0.001$
-Brash (BR): $SM = 1.00 + 0.00484(WTD) - 0.000578(WTD^2)$; $R^2 = 0.712$, $p < 0.001$
Uncut (DN): $SM = 1.00 - 0.00225(WTD) - 0.0000854(WTD^2)$; $R^2 = 0.278$, $p < 0.001$
Burnt (FI): $SM = 1.00 - 0.000427(WTD) - 0.000308(WTD^2)$; $R^2 = 0.354$, $p < 0.001$

Soil moisture (SM) was, therefore, derived for each site and each management scenario from these separate regression equations based on summer WTD.

Cranefly predictions (step 3) were derived from these soil moisture predictions based on the wedge-shaped relationship between cranefly trap abundance and soil moisture described in Section 4.5.1 (e.g. **Figure 114**).

The predicted cranefly abundances across each site are shown in **Figure 131** (Nidderdale), **Figure 132** (Mossdale) and **Figure 133** (Whitendale). Overall, as future climate change is predicted to result in lower soil moisture levels, management choices can either buffer against or exacerbate summer drought on the three blanket bog sites. Burning is generally predicted to result in the driest soil and hence the lowest projected cranefly numbers, whereas mowing and leaving brash provide a substantial buffering capacity as cranefly numbers remained fairly stable over time. However, the impact of uncut and mown with brash removed varied much more depending on the overall wetness of the site. Whereas the two drier sites, Nidderdale and Mossdale showed a clear negative impact of brash removal indicating drying out of peat, the wettest site, Mossdale, showed hardly any difference between the two mown managements. Moreover, the future uncut management showed the highest cranefly numbers at Whitendale, second highest at Nidderdale and third highest at Mossdale.

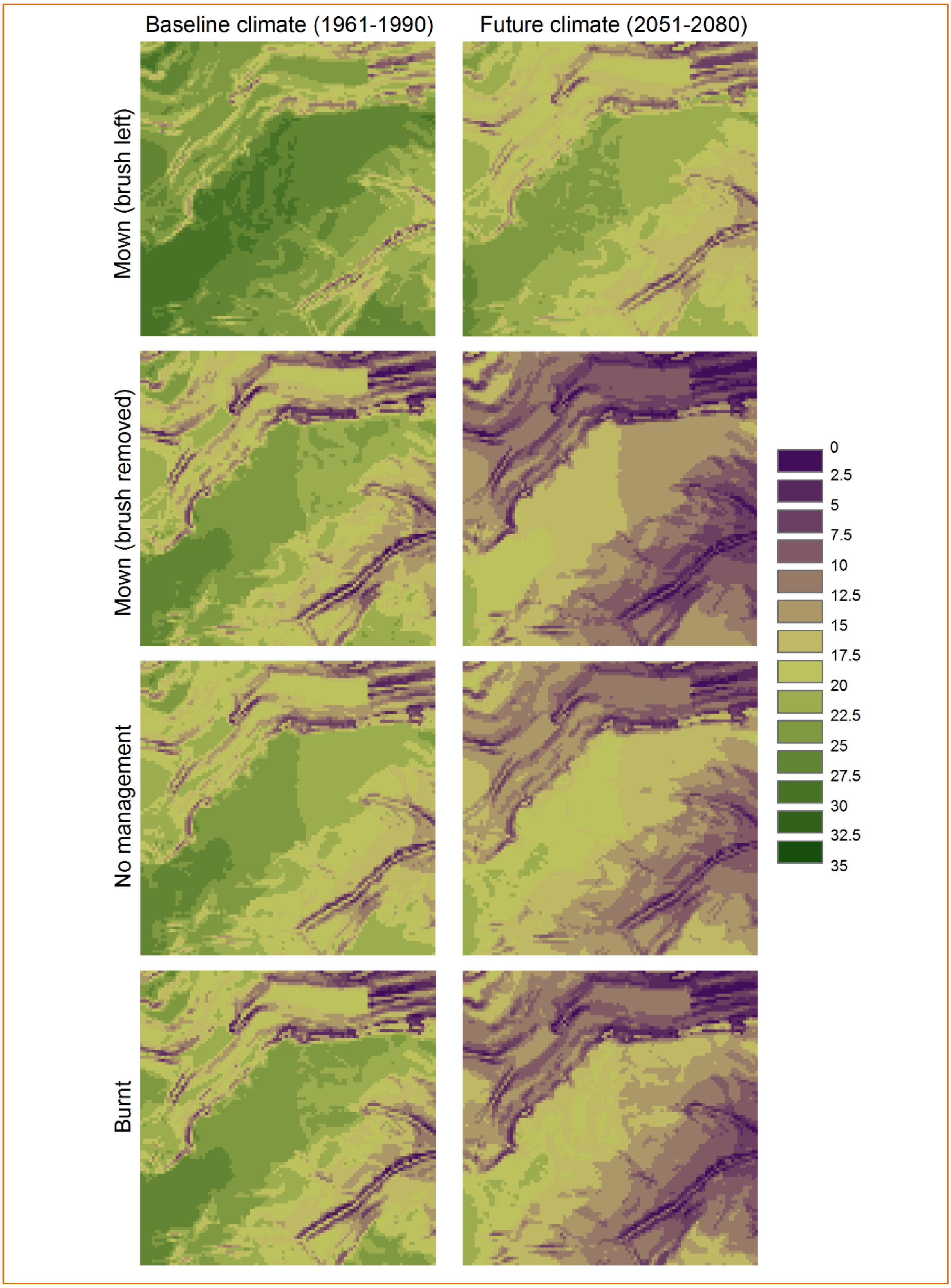


Fig. 131 Projected average total annual cranefly abundance (individuals per m^2) in the 5x5 km square surrounding the **Nidderdale** field site under (**left**) baseline (1961-1990) and (**right**) future (2051-2080) climate scenarios with different methods of habitat management (mowing with brash left, mowing with brash removed, no management (uncut) and burning), modelled as described in the text.

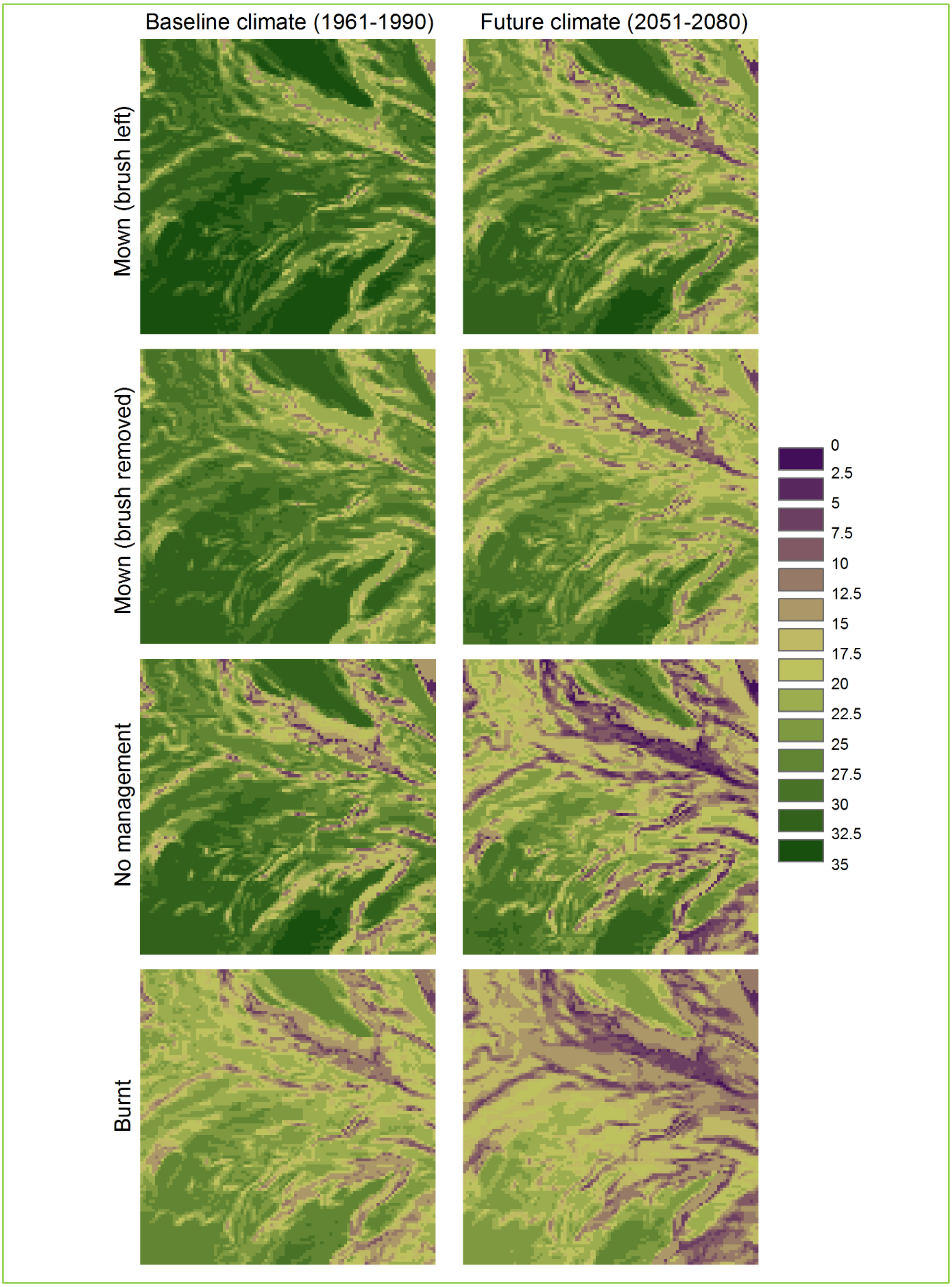


Fig. 132 Projected average total annual cranefly abundance (individuals per m^2) in the 5x5 km square surrounding the **Mossdale** field site under (**left**) baseline (1961-1990) and (**right**) future (2051-2080) climate scenarios with different methods of habitat management (mowing with brash left, mowing with brash removed, no management (uncut) and burning), modelled as described in the text.

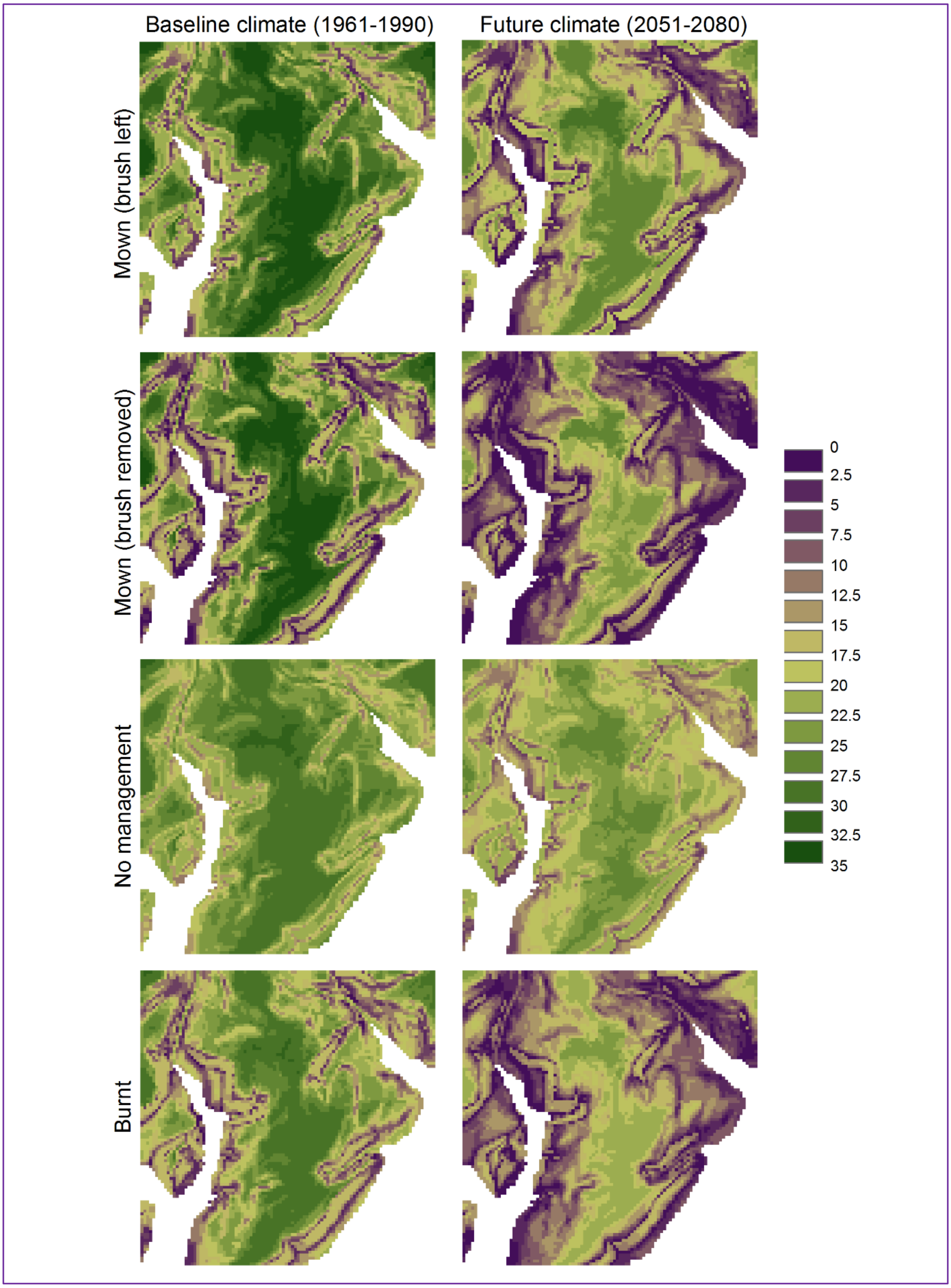


Fig. 133 Projected average total annual cranefly abundance (individuals per m^2) in the 5x5 km square surrounding the **Whitendale** field site under (**left**) baseline (1961-1990) and (**right**) future (2051-2080) climate scenarios with different methods of habitat management (mowing with brash left, mowing with brash removed, no management (uncut) and burning), modelled as described in the text. White areas are those under 250 m a.s.l. which do not support the moorland habitat that is the focus of this study.

A more detailed comparison between sites indicated that climate change would result in a reduction in average annual cranefly abundance (**Figures 131-133**; across all managements) from 24.9 to 20.0 (-19.7%) individuals per m^2 in Mossdale, 19.8 to 14.0 (-29.3%) in Nidderdale and 17.5 to 11.4 (-34.9%) in Whitendale. However, if the Mossdale landscape was managed by mowing, with brash being left in place, the reduction in average annual cranefly abundance would be only from 28.6 to 23.9 (-16.4%) individuals per m^2; if it was mown and brash removed the reduction would still only be from 26.4 to 22.0 (-16.7%); if it was unmanaged the reduction would be from 24.6 to 18.4 (-25.2%); and management by burning would lead to a reduction from 20.1 to 15.6 (-22.4%). If the Nidderdale landscape was managed by mowing, with brash being left in place, the reduction in average annual cranefly abundance would only be from 23.4 to 18.6 (-20.5%) individuals per m^2; if it was mown and brash removed the reduction would be from 18.3 to 11.6 (-36.6%); if it was unmanaged the reduction would be from 18.9 to 13.5 (-28.6%); and management by burning would lead to a reduction from 18.7 to 12.2 (-34.8%). If the Whitendale landscape was managed by mowing, with brash being left in place, the reduction in average annual Cranefly abundance would still be from 19.3 to 12.8 (-33.7%) individuals per m^2; if it was mown and brash removed the reduction would be from 16.3 to 8.4 (-48.5%); if it was unmanaged the reduction would be from 19.1 to 15.0 (-21.5%); and management by burning would lead to a reduction from 15.3 to 9.2 (-39.9%).

In conclusion, climate change is projected to lead to lower summer water table depth in upland moorland landscapes and hence reduced soil moisture and consequently a reduction in cranefly abundance in the spring, with important potential consequences for upland bird species which are reliant on craneflies to feed their chicks. Moorland management, however, has the potential to ameliorate some of this effect; in particular mowing and leaving brash in place rather than burning may increase water table depth (i.e. wetter). However, so far no consideration has been given to the predicted increase in winter precipitation, potentially causing water logging and drowning of cranefly larvae as indicated by the observed relationship of emergence versus soil moisture (e.g. **Figure 114**). Therefore, mowing might not always be the better management tool for cranefly (and likely therefore also bird) numbers compared to burning when considering site wetness. However, it is important to note that these predictions are based on the observed site specific relationship between soil moisture and WTD, which lacked representation at the drier end (**Figure 130**), particularly important for the climate predictions. Therefore, longer monitoring over anticipated future dry spells would enable a more robust prediction. Particularly 2015 and 2016 were very wet, limiting the assessment of impacts at the drier end to mainly one year (2014). Ideally further controlled laboratory studies would also be conducted to specifically assess the upper and lower moisture ranges of craneflies overall and different species (which might differ in this respect).

Predicting management effects on golden plover, dunlin and red grouse abundance

Notwithstanding any potential winter impact, the predicted future summer changes in cranefly abundance were used to project future upland bird abundances based on established relationships between cranefly and bird abundances derived by Carroll et al. (2015) for the southern Pennines. These relationships describe the predicted number of individuals of three bird species (golden plover, dunlin and red grouse) in a 1 × 1 km grid cell for a given average abundance of craneflies (individuals per m^{-2}):

$$\text{Golden plover} = \exp^{(-0.587 + 0.045(\text{abundance}))}$$

$$\text{Dunlin} = \exp^{(-3.873 + 0.086(\text{abundance}))}$$

$$\text{Red grouse} = \exp^{(1.494 + 0.022(\text{abundance}))}$$

These three bird species were selected because they are particularly reliant on craneflies during the breeding season (Carroll et al., 2015). Predictions were based on our 50 × 50 m square grid cell estimates of cranefly abundance (m^{-2}) for each of the four management scenarios under baseline and future climatic conditions (**Figures 131-133**). These cranefly abundance values were averaged to produce mean cranefly abundance for each

of the 25 (1 × 1 km) squares within the three focal landscapes (i.e. Nidderdale, Mossdale and Whitendale). The bird prediction equations (above) were then used to predict the abundance of the three bird species dunlin (**Figure 134**), golden plover (**Figure 135**) and red grouse (**Figure 136**) based on the estimated mean cranefly abundance.

Overall, mean abundance decreased in the order of red grouse to golden plover to dunlin, with highest mean numbers observed at the wettest site Mossdale (7.70; 1.74; 0.19, respectively) and very similar averages for the drier sites, Nidderdale (6.91; 1.35; 0.12, respectively) and Whitendale (6.76; 1.35; 0.13, respectively). In relation to changes in moisture and cranefly numbers, the predicted bird abundances also decreased in the future scenarios across all sites by 44% (dunlin), 22% (golden plover), and 12% (red grouse). Overall, the largest changes, and the lowest numbers overall, reflected those in cranefly numbers (see **Figures 131-133**) and were observed on burnt management; in contrast, uncut and mowing with left brash showed highest predicted bird numbers and reduced losses under future climate change scenarios with increasingly drier summer months.

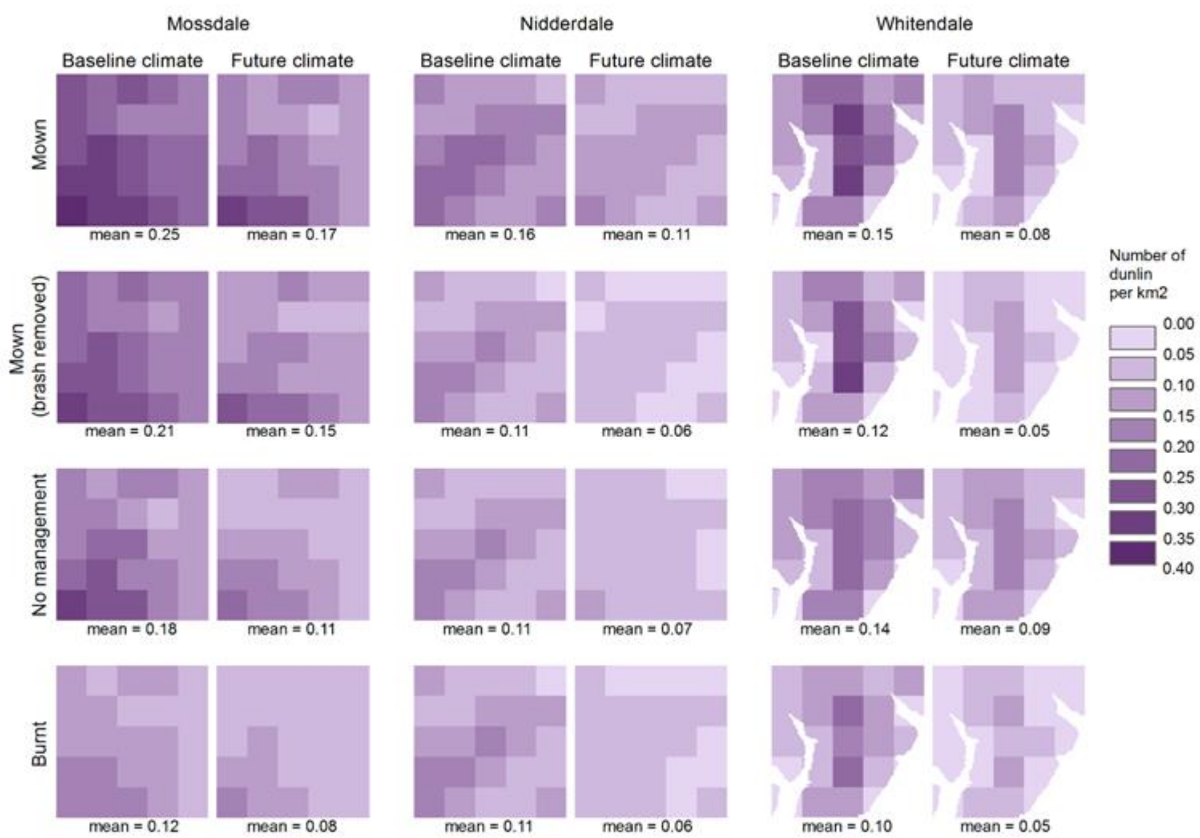


Fig. 134 Projected average annual **dunlin** abundance (individuals per km^2), and the overall mean, within the 5x5 km square surrounding the three sites under baseline (1961-1990) and future (2051-2080) climate scenarios with different methods of habitat management (mowing with brash left, mowing with brash removed, no management/uncut and burning). White areas are those under 250 m a.s.l. that would not support the moorland habitat that is the focus of this study. Predictions are based on the cranefly abundance estimates in **Figures 131-133** and relationship between cranefly abundance and bird abundance described by Carroll et al. (2015).

A more detailed comparison for dunlin abundance between sites (**Figure 134**) indicated that climate change would result in an average reduction in average dunlin abundance from 0.19 to 0.13 (-33%) individuals per km^2 in Mossdale, 0.12 to 0.07 (-38%) in Nidderdale and 0.13 to 0.07 (-48%) in Whitendale. The burning management scenario resulted in the overall lowest numbers under baseline climatic conditions (0.1 km^{-2}) and a reduction in

numbers of 33% resulting from climate change on Mossdale, 45% on Nidderdale and 50% on Whitendale. This compared with the mowing with brash left management scenarios, which had highest baseline dunlin numbers (0.2 km^{-2}) and under which numbers were projected to be reduced by 32% at Mossdale, 31% at Nidderdale and 46% at Whitendale by climate change scenarios.

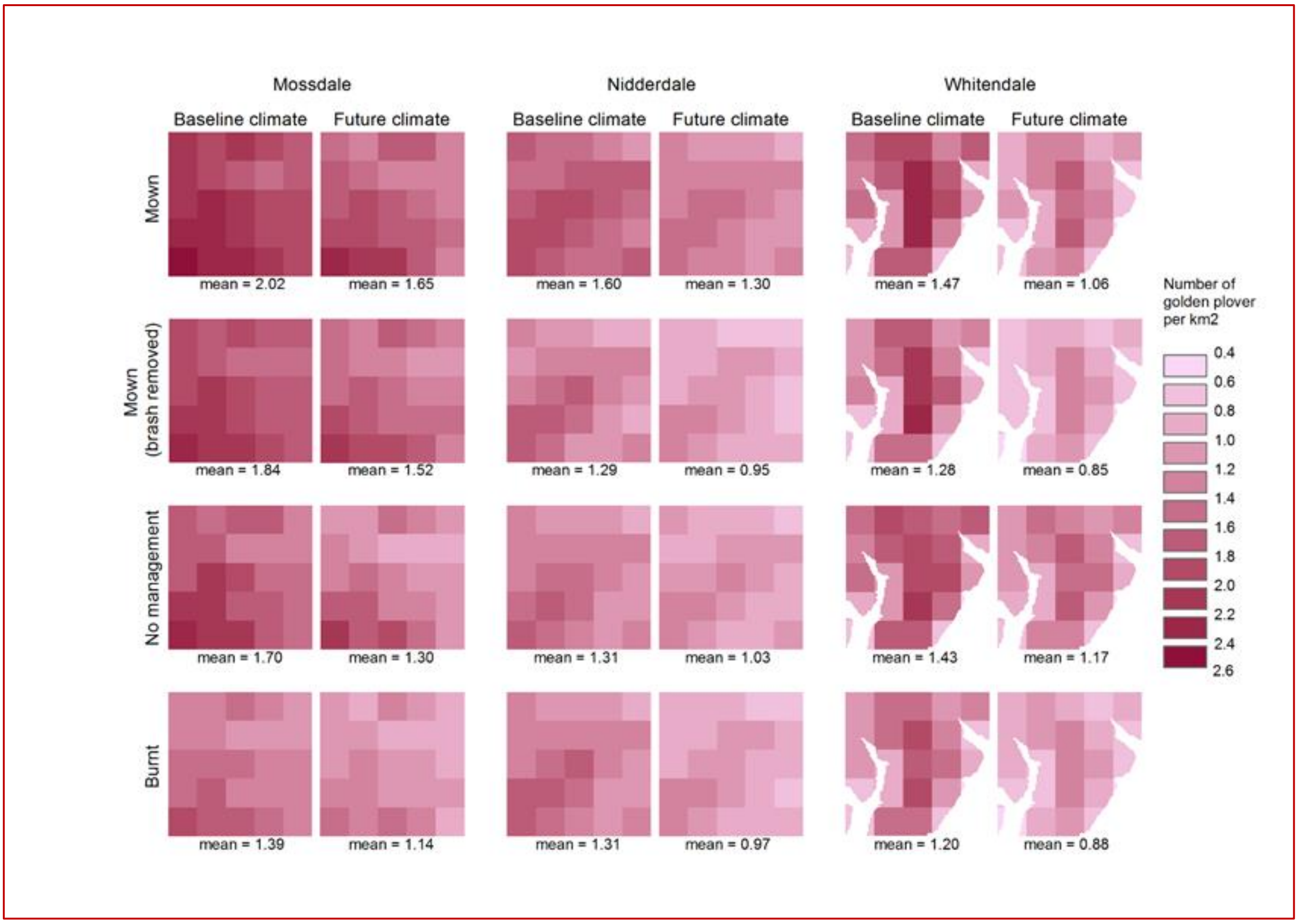


Fig. 135 Projected average annual **golden plover** abundance (individuals per km^2), and the overall mean, within the 5x5 km square surrounding the three sites under baseline (1961-1990) and future (2051-2080) climate scenarios with different methods of habitat management (mowing with brash left, mowing with brash removed, no management/uncut and burning). White areas are those under 250 m a.s.l. that would not support the moorland habitat that is the focus of this study. Predictions are based on the cranefly abundance estimates in **Figures 131-133** and relationship between cranefly abundance and bird abundance described by Carroll et al. (2015).

A more detailed comparison for golden plover abundance between sites (**Figure 135**) indicated that climate change would result in an average reduction in average abundance from 1.74 to 1.40 (-19%) individuals per km^2 in Mossdale, 1.35 to 1.06 (-21%) in Nidderdale and 1.35 to 0.99 (-27%) in Whitendale. Notably, the predictions of breeding golden plover pairs based on cranefly transect numbers for burnt and mown scenarios of just under 0.7 individual per km^2 (0.33 x 2 for a pair) **(Figure 125)** were about half of these predictions. However, this included vegetation height limiting the number of breeding pairs (**Figure 126**); patchiness clearly is important for improving future predictions across the landscape. For the emergence trap predictions considering management (**Figure 135**) the burning management scenario resulted in the overall lowest numbers under baseline climatic conditions (1.3 km^{-2}) and a reduction in golden plover numbers of 18% resulting from climate change on Mossdale, 26% on Nidderdale and 27% on Whitendale. This compared with the mowing with brash left management scenario which had highest baseline numbers (1.7 km^{-2}) and under which numbers were projected to be reduced by 18% at Mossdale, 19% at Nidderdale and 28% at Whitendale by climate change scenarios.

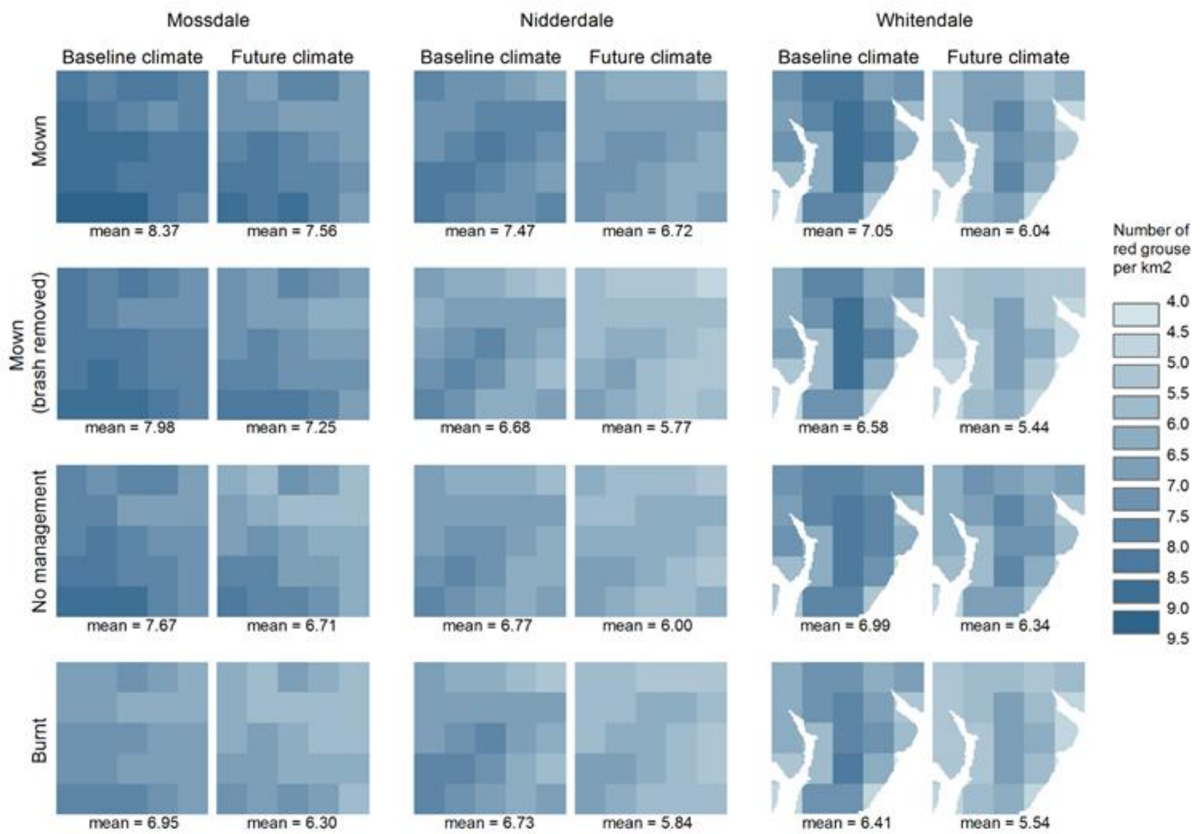


Fig. 136 Projected average annual **red grouse** abundance (individuals per km^2), and the overall mean, within the 5x5 km square surrounding the three sites under baseline (1961-1990) and future (2051-2080) climate scenarios with different methods of habitat management (mowing with brash left, mowing with brash removed, no management/uncut and burning). White areas are those under 250 m a.s.l. that would not support the moorland habitat that is the focus of this study. Predictions are based on the cranefly abundance estimates in **Figures 131-133** and relationship between cranefly abundance and bird abundance described by Carroll et al. (2015).

A more detailed comparison for red grouse abundance between sites (**Figure 136**) indicated that climate change would result in an average reduction in red grouse abundance from 7.70 to 6.96 (-10%) individuals per km^2 in Mossdale, 6.91 to 6.08 (-12%) in Nidderdale and 6.76 to 5.84 (-14%) in Whitendale. The burning management scenario resulted in the overall lowest numbers under baseline climatic conditions (6.7 km^{-2}) and a reduction in numbers of 9% resulting from climate change on Mossdale, 13% on Nidderdale and 14% on Whitendale. This compared with the mowing with brash left management scenario which had highest baseline red grouse numbers (7.6 km^{-2}) and under which numbers were projected to be reduced by 10% at Mossdale, 10% at Nidderdale and 14% at Whitendale by climate change scenarios.

As an *overall conclusion*, the observed differences in soil moisture between managements translated into similar patterns for predicted cranefly and bird numbers. Importantly, all three modelling approaches, two cranefly transect based statistical (golden plover population change and breeding pairs) and one cranefly emergence trap based process level (golden plover, dunlin and red grouse abundance), indicated that mowing can provide some resilience to the predicted increase in future droughts for a key ecological food chain on blanket bog, whereas continued burning practice is likely to aggravate the impacts. However, hydrological conditions during winter, specifically considering predicted increase in precipitation, warrants further investigation considering an observed possible upper soil moisture limitation for cranefly emergence, particularly when considering that mowing

increased site wetness and burning lowered it. Moreover, as impacts differ between years, sites and bird species a more specific consideration of long-term monitoring, site topography and vegetation patchiness could improve future predictions by disentangling specific climate and management impacts. This more detailed view might also include (i) investigating moisture preferences and resilience of different cranefly species as a food source for birds (e.g. changes in climate lead to changes in vegetation and cranefly species, which might or might not vary in their food quality for bird species), and (ii) specifically assessing the upper and lower moisture ranges of survival of cranefly larvae (e.g. laboratory incubation studies). Clearly, understanding where a specific site sits across the soil moisture envelope might then lead to a different decision path on whether to burn or mow, if maximising cranefly numbers (as a food source for specific upland birds) were the main aim.

In summary, the observed effects on cranefly emergence, abundance and model effects on upland bird numbers indicate that:

- Soil moisture was consistently greater under mown than burnt plots. This finding is consistent with the lower water table depth found on burnt than mown plots (Section 4.2.7) and the greater catchment water flows from burnt than mown sub-catchments on two of the sites (Section 4.2.8).
- Cranefly emergence (and estimated abundance based on soil moisture relationships) was significantly higher on mown than burnt plots in the relatively dry summer of 2014. However, there was higher emergence on burnt than mown plots in the wetter summers of 2015 and 2016, although only in the latter year was the effect significant.
- Averaged over the three years, cranefly emergence was greater on mown plots than burnt plots at the two drier sites, but was greater on burnt than mown plots on the wettest (i.e. higher water table depth) of the three sites, Mossdale.
- These observations were consistent with previously reported relationships between soil moisture and cranefly emergence, which suggest that numbers tend to be very low below ~80% and increase sharply with increasing soil moisture up to about a value of 95%, but also suggest a decline under the wettest conditions.
- The modelled implications of these effects in the period 2014-2016 for golden plover fledging production were overall that numbers would be higher in mown than burnt areas, although the numbers of breeding pairs based on vegetation height would be similar.
- This effect was strongest in the relatively dry year of 2014 when cranefly numbers were lowest, and most likely to limit golden plover fledgling numbers.
- MILLENNIA peat development model simulations under future climate change suggest future water table depths would be lower at the two drier sites, but there would be little change at the wetter Mossdale site. This implies that the future area of blanket bog habitat will shrink.
- Modelled abundance of cranefly dependent golden plover, red grouse, and dunlin based on trap emergence data all fell under a climate scenario for 2050-80, with drier and warmer summers.
- The modelled fall in numbers was greater under a burn management than in uncut areas, or in those areas mown with brash left. Hence, mowing (particularly with leaving brash) may enhance the resilience of upland bird populations to the predicted effects of climate change.
- However, this positive effect of mowing in drier areas could lead to declining cranefly numbers (and thus upland breeding birds that rely on invertebrates, especially craneflies, such as golden plover, dunlin and red grouse) in already very wet areas, resulting in burning to be considered as a suitable management tool alongside mowing to prevent increasing wetness above the upper moisture threshold for craneflies.

4.6 *Landscape scale model scenarios*

In the UK, blanket bogs represent about 90% of all peatlands (Bain et al., 2011). Blanket bogs have accumulated peat over about 10,000 years of varying climate, but current bioclimatic models highlight the threat by climate change to their "natural range" (Gallego-Salla et al., 2010), suggesting that they might start to degrade as the climate warms (Gallego-Sala & Prentice, 2013). However, existing bioclimatic models define the range based on current conditions and do not consider extremely important autogenic negative process-level feedbacks within peatlands that may actually act as a 'buffer' to climate change (Swindles et al., 2012). Such feedbacks include ecohydrological feedbacks between changes in water tables depth (WTD) and shifts in vegetation communities (with different rooting depth and litter quality and thus affecting carbon inputs across depth and peat decomposition rates). A better understanding of climate-peatland soil organic carbon (SOC) feedbacks is clearly needed (Davidson & Janssens, 2006) since the mineralisation of peatland soil organic matter (SOM) has the potential to release vast amounts of previously locked-up C into the atmosphere, exacerbating climate change, and affecting overall greenhouse gas (GHG) emissions. Model predictions of peat carbon stock and flux changes rely on capturing seasonal and inter-annual WTD changes. Whereas in the short-term site measurements can be used for model validations, validation over longer-time scales relies on comparing model predictions to WTD reconstructions, for example based on *testate amoebae* (TA). Peat cores provide a stratigraphic (i.e. temporal) archive of past TA species composition, which can be used to predict past moisture conditions (and likely water tables) from an understanding of the contemporary ecology of TA species (Amesbury et al., 2016). TA based reconstructions of past WTDs can then be compared to process model predictions, providing an important hydrological model validation tool; a good fit between TA and model predicted WTDs supports applying model scenarios to explore past management impacts on peatland functioning and carbon storage.

4.6.1 *Comparison of modelled versus testate amoebae based reconstructions of past water tables*

The MILLENNIA peatland model predicts peat hydrological conditions (i.e. WTD) based on climatic conditions either for annual or monthly time steps. Here we used the annual MILLENNIA version (Heinemeyer et al., 2010) for long-term peat accumulation during the Holocene and for the period until 1914, and then either the annual or monthly version (Carroll et al., 2015) until 2012. This choice of model application reflects the availability of climate data availability; see Appendix 15 for further details. We used available reconstructed Holocene climate data (based on a combination of recent instrumental data and a variety of existing multiproxy reconstructions, see Morris et al., 2015) to model long-term peat accumulation, Met Office 5 km gridded data (Perry & Hollis, 2005) for the recent past (1914-1991) and ECN data (ECN Data Centre: http://data.ecn.ac.uk accessed in 2013) for recent present periods (1992-2012). Met Office data were adjusted for elevation for the Moor House NNR site in order to achieve the same long-term average temperature and rainfall amounts as the ECN data (see Carroll et al., 2015). The WTDs predicted by the MILLENNIA model allowed comparison to TA-based WTD reconstructions for a peat core at Moor House (Swindles, unpublished) using the transfer function of Turner et al. (2014).

Water table reconstructions for the Moor House core, based on the TA transfer function, showed an overall good correlation with the MILLENNIA modelled WTD (**Figure 137**; left) although the TA predictions were consistently drier (see Heinemeyer & Swindels, 2018). An offset correction based on the fitted relationship was then used to compare data for the last century. The available ECN climate data (1931-2010) for Moor House were used to make comparisons between annual mean WTD predicted by the MILLENNIA model to the paired years (±1 year) from the Moor House core with the offset corrected available TA-derived WTD (**Figure 137**; right). Not only did the WTD range predicted by the model agree with the TA-derived WTD, but it also reproduced the general pattern of dry versus wet years (MILLENNIA WTD = TA_WTD + 0.004; r^2 = 0.68). Both WTD predictions indicated that the years between 1965 and 1995 were an extended period of fairly dry conditions compared to the long term WTD average of 4.4 ± 1.8 cm (based on annual averages of ECN dipwell data)

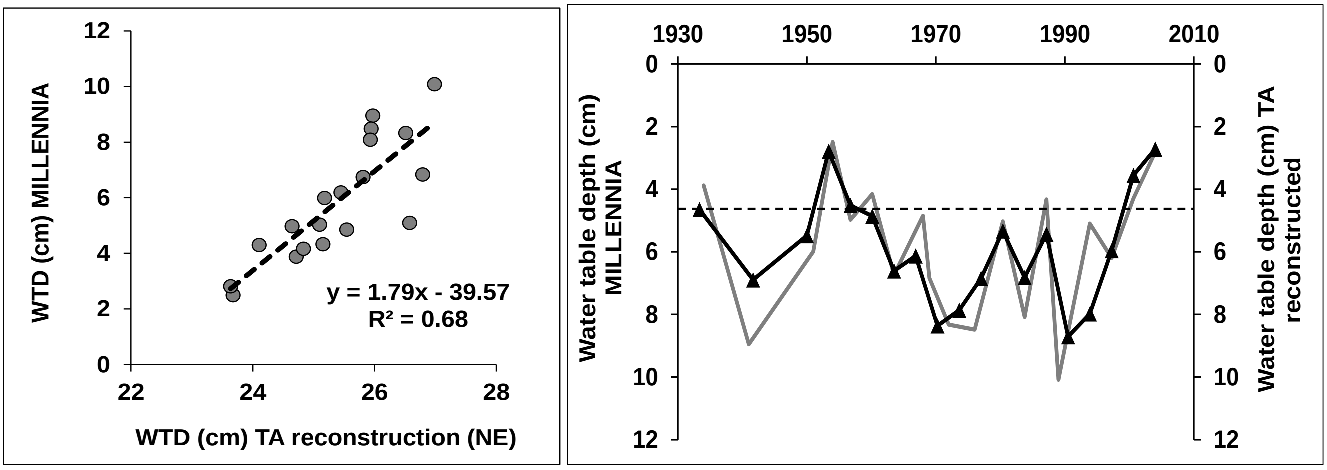


Fig. 137 Left: Linear regression of MILLENNIA model predicted mean annual water table depth (WTD) versus testate amoebae (TA) based predictions (Swindles, unpublished) for a blanket bog at Moor House (NNR) based on an uncorrected transfer function derived for northern England (NE) as in Turner et al. (2014). **Right**: Annual WTD predictions between 1931-2004 from the MILLENNIA model for Moor House shown as 4-year averages (grey line) versus paired years (triangles) of 4-yearly testate amoebae (TA) based predictions (black line) using the offset regression (**left**) for TA-based WTD corrections. The dashed line indicates the long-term (1995-2012) mean annual WTD at Moor House (ECN data licence: ECN:AH2/14).

The annual MILLENNIA model prediction of WTD during 1931-2012 also showed very good agreement with the about 4-yearly offset corrected TA-derived WTD (see **Figure A15.3** in Appendix 15), capturing peaks and troughs as well as the average trend. Moreover, the model predicted WTD (WTD_{Mod}) followed the available annual ECN site WTD (WTD_{ECN}) data very well ($WTD_{ECN} = WTD_{Mod}*0.813 + 0.595$, $r^2 = 0.57$; see Carroll et al., 2015).

4.6.2 *Peat model grouse moor management scenarios*

Based on the above historic WTD validation and in order to highlight the importance of site history (i.e. past management) in determining current C sink strength, a further model study was conducted for Moor House (see Heinemeyer & Swindels, 2018), which was a shooting estate between 1842 and 1951. The grouse moor management scenarios reflected available site information (Rob Rose, CEH; personal communication), which indicated a 20 year burn rotation, with assumed associated drainage impacts from 1830, before intensification of grouse shooting (to enable enhanced heather growth and drier access conditions for gamekeepers), until the late 1950s. Burning was assumed to have started in 1850 and to reduce net primary productivity (NPP) to 1% in the burnt year (with charcoal adding about 5% to an inert carbon pool; Section 4.4.4), subsequently recovering in a sigmoidal shape to 100% by either 5 or 10 years after burning (based on Heinemeyer et al., unpublished data). Drainage was assumed to lower water table depths on average by 5 cm, based on the field evidence of Wilson et al. (2010). Increased (lower) WTDs were assumed to enhance decomposition and increase the associated CO_2 fluxes, but decrease CH_4 emissions, similarly to previously modelled impacts of natural WTD changes (Heinemeyer et al., 2010). Drainage (grip) effectiveness was assumed to be at optimum for 25 years and declining to 60% over the subsequent 15 years (renewed once in 1871 and then maintained at optimum until 1905, reflecting intense grouse shooting), finally declining to 0% by 1955. Grazing pressure was assumed to be insignificant above 450 m (i.e. to cause no reduction in NPP at the modelled altitude of 550 m a.s.l.).

Use of the available long-term climate data for a nearby Northern England peatland site (Morris et al., 2015), adjusted to the long-term mean temperature and total rainfall for Moor House, together with Moor House ECN climate data since 1931, provided a basis for a comparison of the WTD predicted by the MILLENNIA model over an extended time period (1750s till 2012), for which TA WTD reconstructions were available (**Figure 138**). Although the general WTD patterns agreed very well between the two predictions, the (unadjusted) TA WTD

predictions were much drier than the MILLENNIA predictions. Whilst MILLENNIA model predictions indicated mean annual WTD conditions during 1760-1830 of between 2-6 cm, the unadjusted TA predictions of WTDs were around 15 to 25 cm. Adjusting for the 1931-2004 determined TA-WTD offset (**Figure 137**) increased the TA-based WTD predictions during 1750 till 1825 to well above the peat surface (annual mean WTD -2.92 ± 6.16 cm; data not shown). However, whereas MILLENNIA model predictions of the unmanaged (no shoot) scenario remained very wet during 1830-1940, unadjusted TA-predictions showed much lower WTD during that period, but overlapped again with the MILLENNIA model predictions from the 1940s onwards.

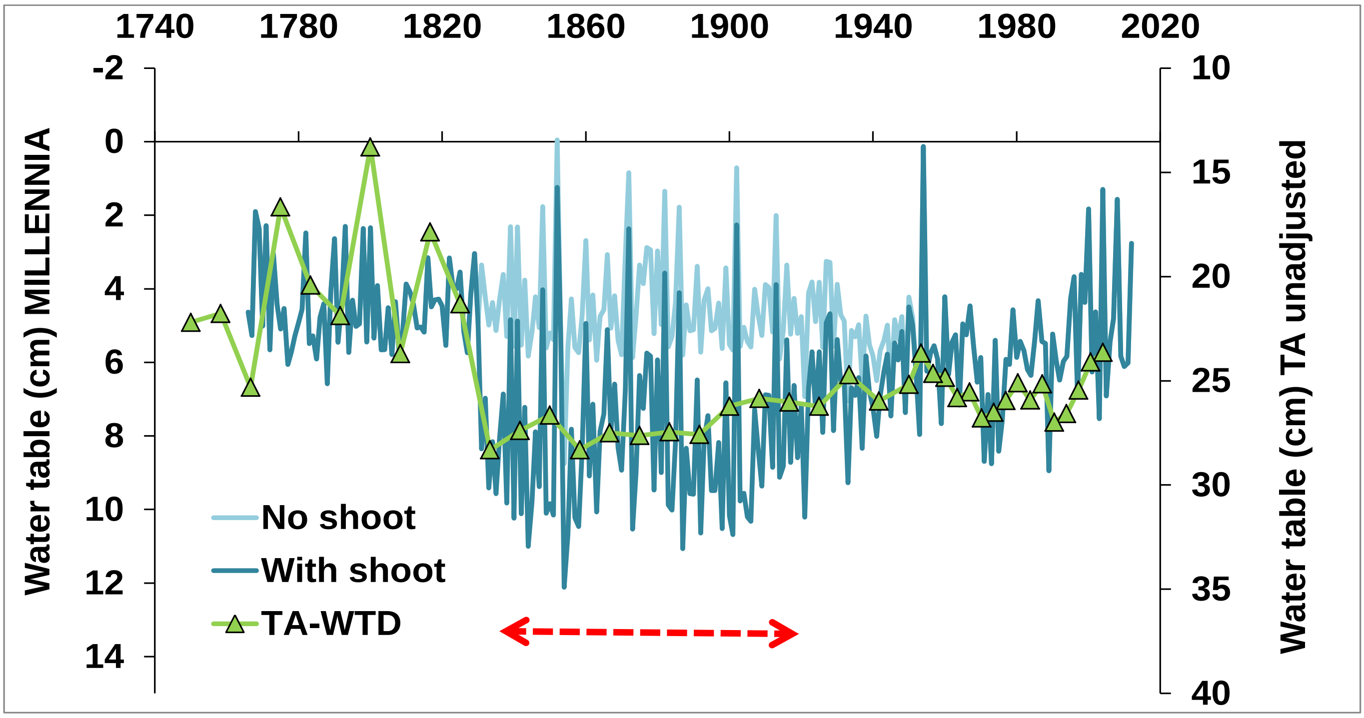


Fig. 138 Mean annual water table depth (WTD) predictions for Moor House (NNR) from the MILLENNIA model (blue lines) versus 4-yearly *testate amoebae* (TA) based WTD predictions (green line; Swindles, unpublished). Light blue indicates the model scenario without grouse shoot management, whereas dark blue indicates the scenario with ~5 cm lower WTD during the grouse moor shoot management period with active drainage (dashed red arrow range: 1830s – 1920s). Note the different y-axis scale (i.e. unadjusted TA predictions).

MILLENNIA model predictions which included a scenario of grouse moor burning, and including active peat drainage during 1830-1920s, with subsequent diminishing drainage until 1950 reflecting naturally infilling of ditches, resulted in a significant reduction in WTD reduction during the period 1830-1930, which was comparable to the unadjusted TA-based record. This is the first study (i.e. Heinemeyer & Swindels, 2018) to reveal a past management impact on peatland functioning based on a comparison of modelled WTD to TA-based model reconstructions. It highlights the need to consider site management history and peat development in understanding paleoecological records and current carbon stock estimates.

The MILLENNIA grouse moor management model scenario period of 1851-1950 (see **Figure A15.4** in Appendix 15) showed a positive annual carbon balance (gain) of 11.7 ± 22.7 g C m^{-2} for the unmanaged scenario (when the mean WTD was 4.8 ± 1.9 cm; **Figure 138**) compared to a negative (loss) balance of -38.6 ± 110.1 g C m^{-2} for the grouse managed scenario with drainage (mean WTD of 7.7 ± 2.1 cm), assuming a 10-year NPP recovery time after burning. The corresponding peat increments predicted by the MILLENNIA model showed a peat growth of 0.03 ± 0.59 cm for the unmanaged versus a loss of -0.06 ± 0.61 cm for the grouse managed scenario (data not shown). Moreover, a 5-year NPP re-growth scenario (data not shown) resulted in an about 45% lower carbon balance loss (-21.6 ± 93.7 g C m^{-2}) and half the peat increment loss (-0.03 ± 0.63 cm) compared to the 10 year NPP recovery. During 1831-1850, the period of drainage only (i.e. no burning), the grouse managed (i.e. drained) scenario reduced the annual carbon balance by 27.8 g C m^{-2} to -7.2 ± 15.8 g C m^{-2} (see **Figure 15.4** in Appendix 15)

corresponding to an annual peat increment reduction of 0.06 cm to -0.02 ± 0.63 cm compared to the unmanaged scenario (i.e. no drainage), reflecting an average WTD reduction by 4 cm to 8.4 cm (**Figure 138**). The no drain 10-year NPP re-growth burn scenario (data not shown) resulted in an about 30% reduced loss of the annual carbon balance of -28.4 ± 108.7 g C m^{-2} and -0.04 ± 0.63 cm peat increment loss (mean WTD of 4.8 ± 1.9 cm) compared to only -10.9 ± 92.6 g C m^{-2} and -0.01 ± 0.64 cm, respectively, for the 5-year NPP re-growth burn scenario (i.e. 80% less carbon and peat increment loss compared to the default burn and drain scenario). Overall, grouse moor management scenarios predicted increased (lower) WTD by drainage and reduced C inputs through burning including lower NPP following burning.

The effect of grouse moor management model scenarios on the annual carbon balance, and hence peat, reflected changes in the carbon fluxes including methane (CH_4; data not shown but see Heinemeyer & Swindels, 2018). Overall, drained and 10-year NPP recovery scenarios reduced both mean annual carbon losses from soil CO_2 fluxes (308 ± 106 g C m^{-2}) and annual net CH_4 emissions (4.9 ± 8.3 g C m^{-2}) compared to unmanaged scenarios for which values were 417 ± 60 g C m^{-2} for CO_2 and 13.2 ± 20.4 g C m^{-2} for CH_4 emissions. The changes in CH_4 emissions reflect alterations in methane oxidation and emissions including plant mediated transfer (PMT) processes via sedge leaves and stems. However, whereas no drain 10-year NPP burn scenario decreased CO_2 (292 ± 110 g C m^{-2}) and increased CH_4 (9.8 ± 18.0 g C m^{-2}) emissions, the 5-year NPP scenario resulted in both increased CO_2 (348 ± 92 g C m^{-2}) and increased CH_4 (11.2 ± 18.8 g C m^{-2}) emissions, reflecting quicker vegetation regrowth and thus NPP and PMT recovery. These MILLENNIA model predicted mean annual soil respiration and net methane emissions were of the same order of magnitude as the modelled mean soil respiration (**Table 15**) and estimated median CH_4 emissions (**Table 16**) based on chamber fluxes.

The model grouse moor management scenario predicted reductions in carbon budgets agree with less peat accumulation on burnt compared to unburnt plots as reported for Moor House by Garnett et al. (2000). However, the model did not account for any potential charcoal impacts on hydrology and decomposition (Section 4.4.4). In particular, impacts on bulk density (see Section 4.3.3) with possibly impacts on water content and effects of negative priming (Lu et al. 2014) on decomposition rates (Section 4.4.2) should be considered in future model developments.

4.6.3 *Peat model past land management scenarios*

A final model study was performed to consider the implications for soil organic carbon (SOC) storage of potential grouse moor management (burn and drain) in a wider historical and land use management context as well as highlighting the potential of SOC sequestration under a no management model scenario (for which there are generally no site data available). This study aimed to develop a modelling tool to allow predictions of SOC storage across wider catchment areas, including identifying areas likely to have been peatlands in the past (and as such providing additional guidance as to where to prioritise restoration efforts as past losses would equate to potential future C gains). Notwithstanding uncertainties in representing specific management impacts on soil carbon accumulation, there is plenty of evidence that historic human land use altered soil carbon cycling. In particular, past (i.e. Holocene) agricultural management will have changed landscape carbon stocks, which is as yet not accurately represented in global C-cycle models (Stocker et al., 2018). For example, the recent Defra funded study on agricultural impacts on lowland peatlands reported a mean loss of 691 g C m^{-2} yr^{-1} (Evans et al., 2017b) under various crops (or a total loss of 2.3 kgC m^{-2} over 3.5 years of measurements); the main factor identified was water table drawdown via drainage, but possible additional (wind) erosion could increase this C loss even further. It was predicted that that many of these lowland agricultural peatlands will cease to exist within a few centuries or even less should the currently observed rates of carbon loss continue (Evans et al., 2017b). Most importantly, large areas of peatlands may have disappeared in the past due to use of peat for fuel, as described in an unpublished PhD thesis (Ardron, 1977).

Surprisingly, this human induced peat loss has never been included in any carbon stock modelling study or assessments of potential peat areas. For all three sites in this study, there is anecdotal evidence (from personal communication with farmers) and visual evidence (from the presence of peat banks) for localised peat cutting until the 1950s, as well as agricultural management of peat areas. In fact, the impact on the landscape will likely have been much larger than suggested from these two sources of evidence, as human management and peat use is likely to have continued for hundreds, if not thousands, of years. In particular, before large-scale extraction of fossil fuels, peat would have been a major and easily accessible fuel in many areas of the UK and Europe, for both home and industrial use, as shown for the Peak District and other UK areas by Ardron (1977); this could have resulted in major changes for former peatland areas. However, currently models tend to predict soil carbon dynamics based on currently observed soil types and soil carbon stocks; where peat has disappeared this would result in inadequate assumptions (i.e. mineral soils of low SOC content). Therefore, modelling future carbon stocks based on tuning SOC processes in C-cycle models to simulate currently observed SOC stocks might be misleading and underestimate the climate change mitigation potential of soils.

Whereas mineral models such as Century (Parton et al., 1983) are commonly used only for mineral soils, organic soil models such as MILLENNIA (Heinemeyer et al., 2010) are applied only in a peatland context. However, as soil decomposition is mostly based on generic litter decomposition processes, it should be possible to apply these models in a more general context to predict soil carbon dynamics. Whereas Chimner et al. (2002) applied the Century model to peatlands, so far no peatland model has been applied outside its specific peat context. Although peatland model predictions of potential peat depth would only apply to peatland areas and would become meaningless elsewhere, their SOC stock predictions based on globally derived climate relationships with litter input and decomposition should still apply. Therefore, the MILLENNIA model was adapted to represent management impacts on carbon cycle processes and the resulting carbon stocks. This allowed a comparison of an unmanaged scenario predicting potential SOC accumulation to a likely managed scenario resulting in losses from this potential long-term sequestration, which could then be related to currently estimated carbon stocks (i.e. the National Soil Resources Institute's (NSRI) SOC data are up-scaled estimates (see Bradley et al., 2005) based on actual SOC data collected from a 5 km sample grid under random soil type and land use (i.e. look up table data are used to estimate an average SOC for each 1 km grid), but data do not take into account variations due to differences in elevation, slope and aspect). A model-data comparison for the Moor House (NNR) bog area by Heinemeyer et al. (2010) highlighted the importance of including topographic conditions in SOC estimates and model comparisons. Therefore, NSRI data should not be used as 'real' SOC stocks to be compared to modelled SOC data as part of a model validation.

Figure 139 provides an overview of the MILLENNIA peat cohort model structure, which predicts carbon input from net primary productivity (NPP) and losses from decomposition and the here considered additional human impacts. The major human impacts are related to affecting either carbon stocks through fuel usage (peat cutting for burning) or decomposition processes via agricultural activity (drainage and fertiliser additions) or carbon inputs via NPP (grazing or harvest). These were modelled to change over time, as populations grew. In addition the model also captured the historical change in climate and the topographic conditions across the landscape affecting runoff and erosion. **Table A15.1** (in Appendix 15) provides a detailed overview of the managements that were modelled (i.e. burning, grazing, harvest, peat cutting, ploughing, fertilising), their constraints (time and topographical) and their likely impacts on specific model processes. In addition some natural disturbance is assumed, in the form of natural fires every 200 years. An assumed peat cutting depth of 40 cm was translated into an overall peat volume and then into an average annual long-term cutting depth impact across the landscape (of around 0.4 cm by 1800), which depended on population density and therefore demand change over time and the total area suitable for peat cutting. Finally, burning, grazing and harvest, as well as peat cutting, were modelled to reduce NPP inputs, with a subsequent recovery period.

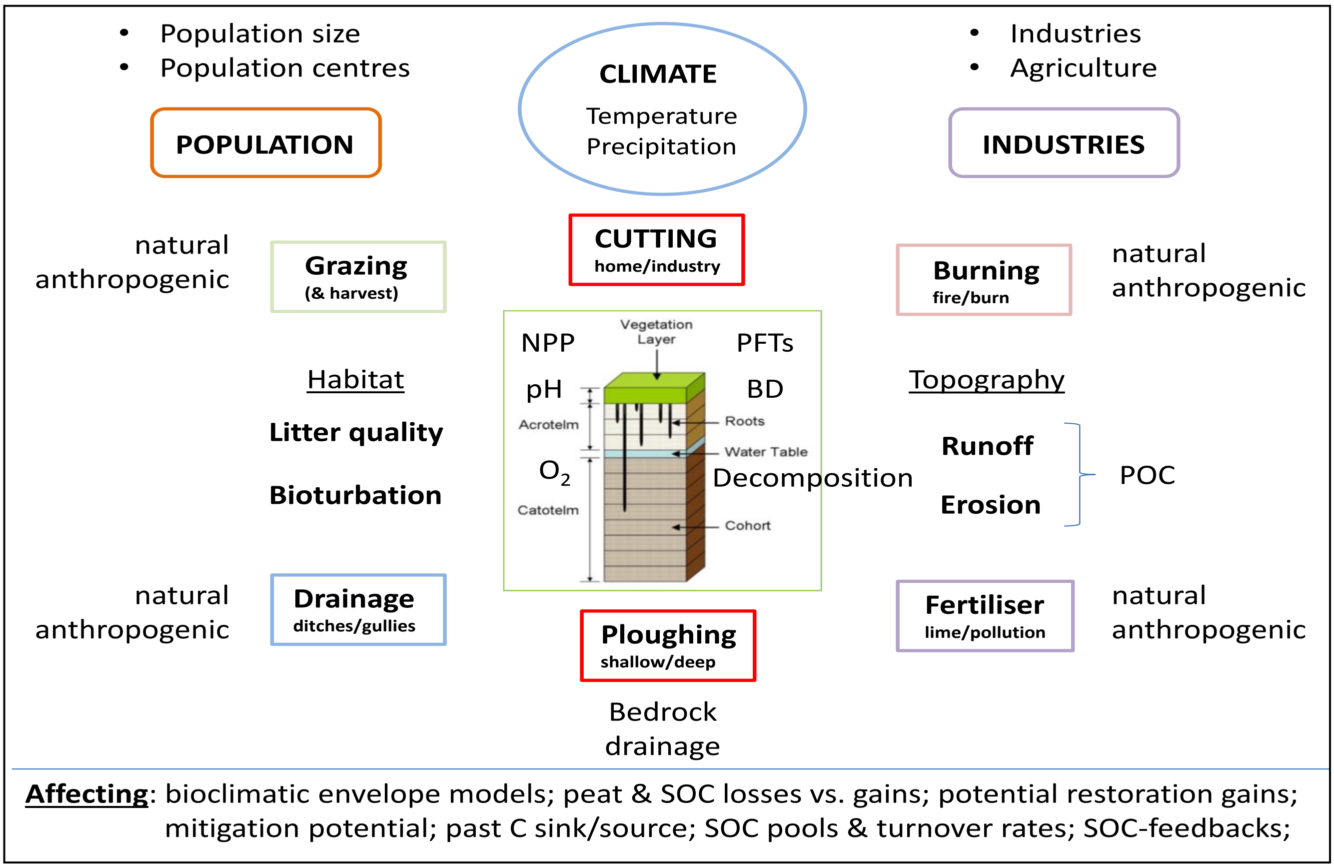


Fig. 139 Schematic concept of land management processes affecting peatland development and carbon accumulation in addition to natural processes such as climate, natural fires, grazing and bedrock drainage. The main human-regulated processes are grazing, cutting, burning, drainage and fertiliser applications together with land cultivation (ploughing), with process intensity being proportional to population density and industrial demand. Resulting carbon and peat accumulation processes affect both interpreting and predicting current and future soil organic carbon (SOC) stock changes and associated feedbacks. The central picture shows the MILLENNIA model (Heinemeyer et al., 2010) cohort structure including topography affecting runoff and erosion (and thus particulate organic carbon, POC), and habitat affecting net primary productivity (NPP) inputs and decomposition via oxygen (O_2) availability in relation to water table depth, pH, bulk density (BD) and plant functional types (PFTs) affecting rooting depth and litter quality.

Whereas for some aspects, literature was available, estimates of the impact of managements were based primarily on non-peer-reviewed “grey” literature (Appendix 15 provides details of sources). However, the model scenarios are intended to only provide a first attempt in testing the hypothesis that potential (i.e. unmanaged) peat carbon stocks are much larger than anticipated by current (i.e. managed) stocks. This would then help to place any current peatland management and restoration effects on soil carbon stocks into the context of past and potential future changes across peatland areas and elsewhere. Model scenarios were run over 8,000 years, reflecting the general onset of significant soil carbon accumulation in peatlands across the UK (e.g. Tallis, 1991, 1998). Adjusted climate data were used, as in the previous section (4.6.2) for Moor House NNR, but were adjusted for the respective site location’s long-term average climate (obtained from the UK’s Met Office website).

For a test scenario at Moor House, individual managements resulted in very different impacts on SOC stocks (see **Figure A15.5** in Appendix 15). Overall, all managements reduced SOC, with peat cutting and burn management having the largest one-off impact of just under 1 kg C m^{-2}. Ploughing had the second largest impact. The sensitivity of these impacts to various assumptions about management was also explored (see **Figure A15.6** in Appendix 15). These included a one-off cutting depth of 40 cm per location versus no cutting, reduced net primary productivity (NPP) after cutting versus no effect, fire occurrence (i.e. natural fire of every 200 years versus no fires), and ploughing intensity (i.e. three levels of ploughing aeration impact). However, the deep cut was not implemented for the landscape scale predictions as this would not reflect a realistic long-term peat usage impact across the larger landscape.

The effects of the unmanaged and managed scenario were first modelled at a 50 x 50 m resolution for a 5 x 5 km upland area around Poundsgate, near Dartmoor in Devon, UK (**Figure 140**). This area was selected as it covered a range of combination of elevation, slope and aspect and included some small towns. The model scenarios of peat cutting, drainage, harvest, ploughing and fertilisation were all applied considering the limitation criteria described in Appendix 15. The predicted soil carbon stocks in the unmanaged scenario were about 60-80 kg m^{-2} in valleys and above 100 kg m^{-2} on the hill tops (**Figure 140**; top left). The managed scenario indicated a considerable loss of carbon, with large losses in valley areas showing remaining soil carbon amounts of only around 1-20 kg m^{-2} (**Figure 140**; top right). Other high soil carbon loss areas (**Figure 140**; bottom) were the upland areas accessible and suitable for peat cutting (i.e. <450 m elevation and 0-5⁰ slope).

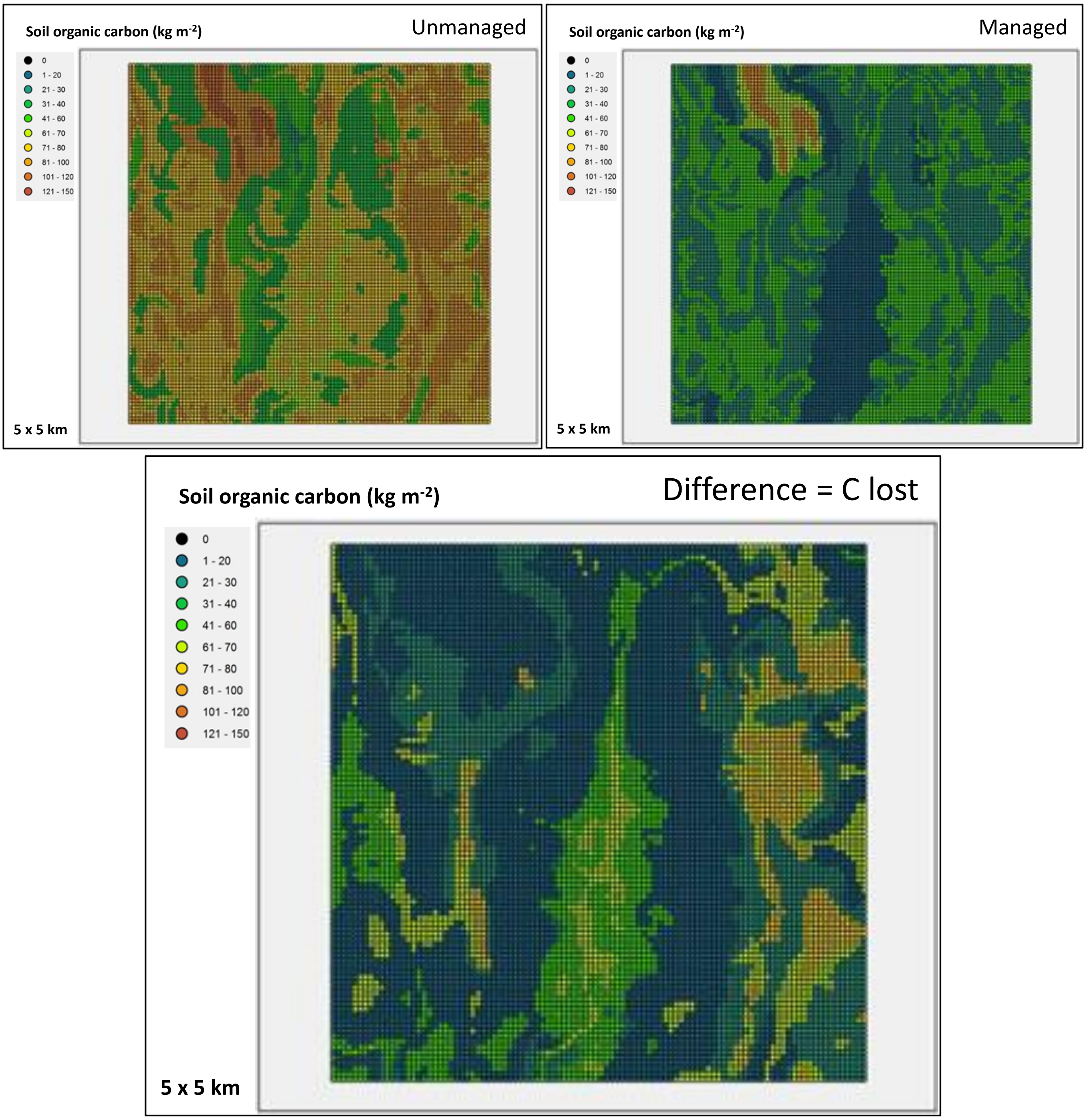


Fig. 140 Comparison of MILLENNIA model (Heinemeyer et al., 2010) predicted scenarios of carbon accumulation over 8,000 years with or without human management impacts since 1100 AD for the 5x5 km area around Poundsgate (Dartmoor, UK). **Table A15.1** in Appendix 15 provides full details of the management scenarios. Shown are the 50x50 m gridded unmanaged model output (**top left**) versus the managed scenario (**top right**) and the corresponding amount of lost soil carbon (**bottom**). The elongated central area is a valley region with steep slopes leading into the surrounding flatter upland areas.

The same model scenarios were then also run for a further five sites across the UK, covering a range from Dartmoor in the south to the Flow Country in Scotland (**Figure 141**). Mean elevations for the five 5 x 5 km areas ranged from York (15 m) to Moor House (560 m), and average climatic conditions ranged from dry and warm at Poundsgate, (10.3°C and 1385 mm) to cold and wet at Moor House (5.2°C and 2000 mm). As current (and therefore likely also historic) population densities for each 25 km^2 area were different, overall population pressure was calculated separately, and was used only to adjust the peat cutting intensity. Other management impacts such as agriculture represented population pressure only by technological advances (i.e. ploughing depth), increased demand across the landscape (i.e. increased range in altitude of drainage but decreased altitude effect on grazing and harvest removal impacts on NPP) and effectiveness (i.e. fertiliser impact). See **Table A15.1** in Appendix 15 for a summary.

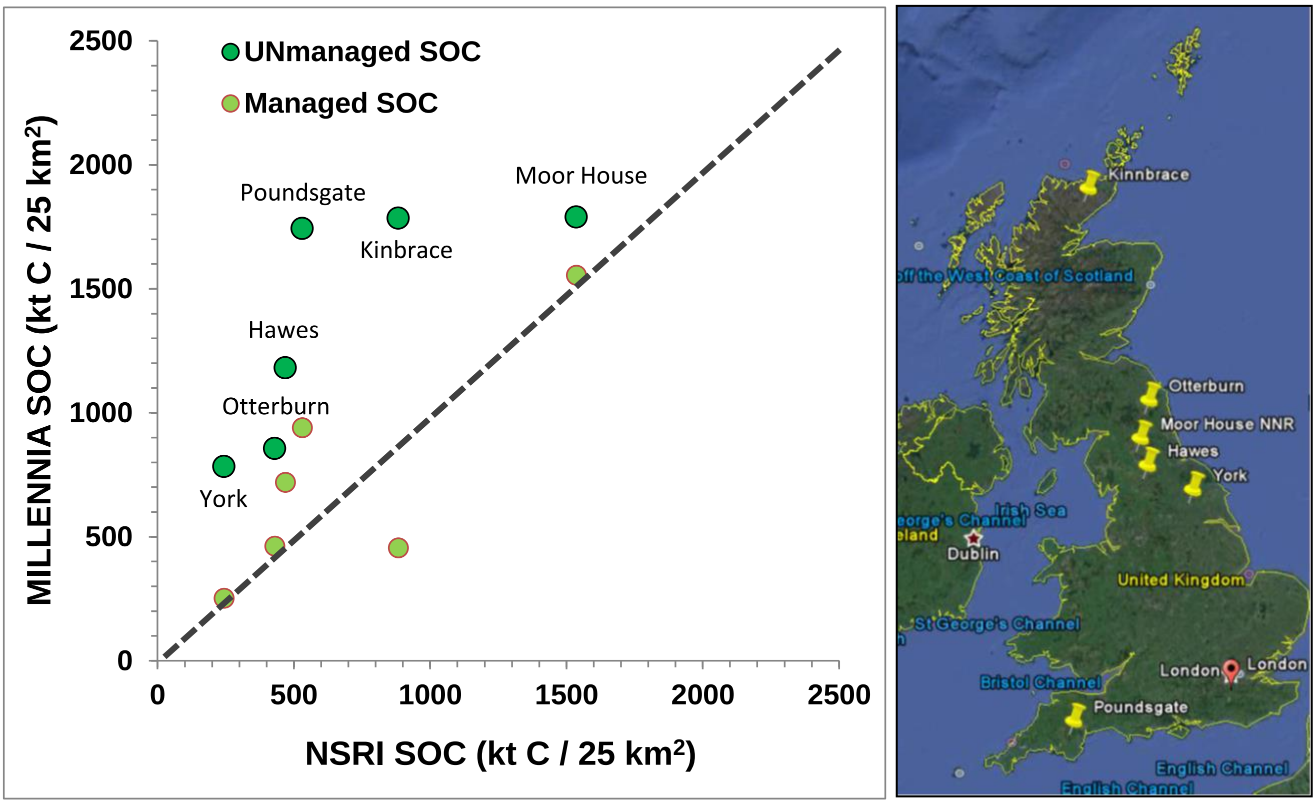
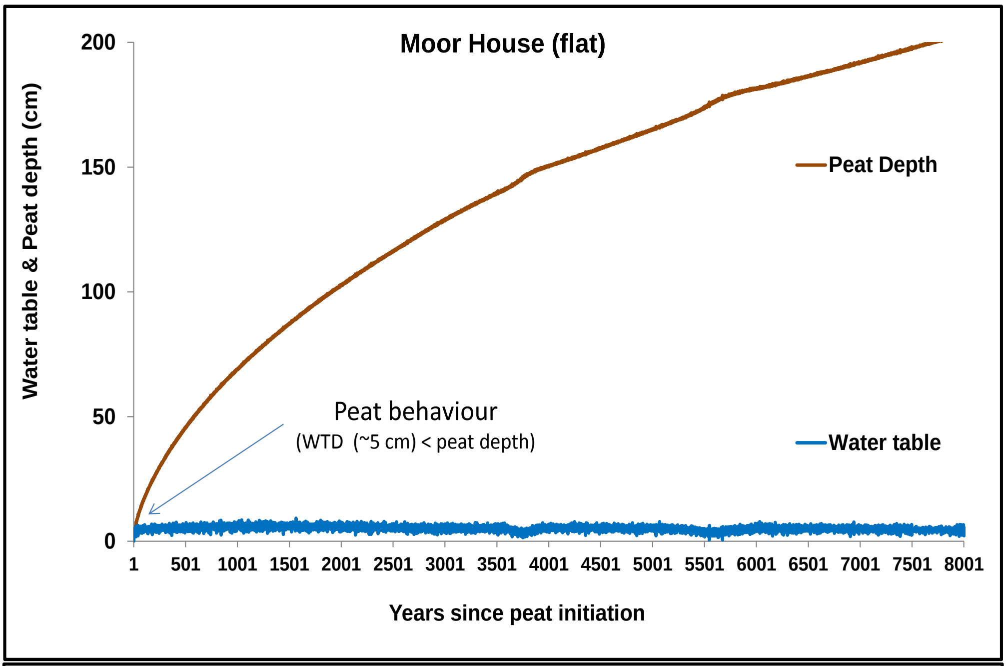


Fig. 141 Comparison of MILLENNIA model (Heinemeyer et al., 2010) predicted soil organic carbon (SOC) accumulation over 8,000 years for the unmanaged and the managed scenario (see **Table A15.1** in Appendix 15) plotted against current values for the same area from the National Soil Resources Institute (NSRI) soil carbon inventory. SOC predictions for both scenarios were up-scaled to the 5x5 km (25 km^2) grids for the six locations, the locations of which are shown on the right hand map.

A comparison of the unmanaged predictions with the estimated SOC stocks in the NSRI soil database revealed much higher values for the unmanaged model predictions at all the six sites (the locations of which are shown in **Figure 141**; right), with the largest differences for Poundsgate, Kinbrace and Hawes. As for Poundsgate (**Figure 140** above), the managed scenario reduced the predicted SOC values across all sites. The largest effect was observed for peatland areas with large population centres coupled to suitable peat cutting and suitable agricultural areas at lower elevations, specifically Poundsgate and Kinbrace but also Hawes and York. The modelled effects of management at Moor House and Otterburn were much lower, reflecting their lower population and higher elevation, making them less suitable for peat cutting and agricultural usage. Remarkably, the points for the managed scenarios fell surprisingly close along the 1:1 line, hence showing good agreement between the managed modelled scenario estimates and the NSRI's SOC stock inventory data (**Figure 141**; left). However, a direct comparison in this context is neither valid nor intended as the NSRI data are not observations but up-scaled estimates (see above NSRI data section). Notably, on average over a 25 km^2 area across the five sites the predicted mean soil carbon stock loss of 625 kt C was nearly equal to the amount remaining (730 kt C) with a mean soil carbon stock prediction of the unmanaged scenario of 1,356 kt C across a 25 km^2 area. Upscaled

to the area of Great Britain of around 209,000 km², this equates to a loss of around 5.2 Gt C; in comparison the current soil carbon stock estimate based on NSRI data to 1 m depth is around 4.1 Gt C (Bradley et al., 2005).

Whereas the model predicted 'organic' soil (i.e. peat), with a WTD within and mostly near the surface of the peat profile, at some sites (for example Moor House; **Figure 142**; top), the model also predicted 'mineral' soil carbon accumulation for locations outside peat areas (i.e. WTD at the bedrock below the dry 'peat' column). Closer inspection of such locations (for example, Otterburn; see **Figure 142**; bottom) sometimes revealed a transition between 'mineral' and 'peat' WTD behaviour, with dips in and out of 'peat' behaviour (reflecting climatic changes), and the onset of such 'peat' behaviour at a peat depth of around 35 cm (**Figure 142**; bottom). Notably, this is very close to the peat depth classifications used across the UK (30 - 50 cm). Therefore, model predictions might provide additional information on obtaining information on long-term C sequestration across larger areas under a no management scenario as well as for restoration measures, which otherwise are not currently available.



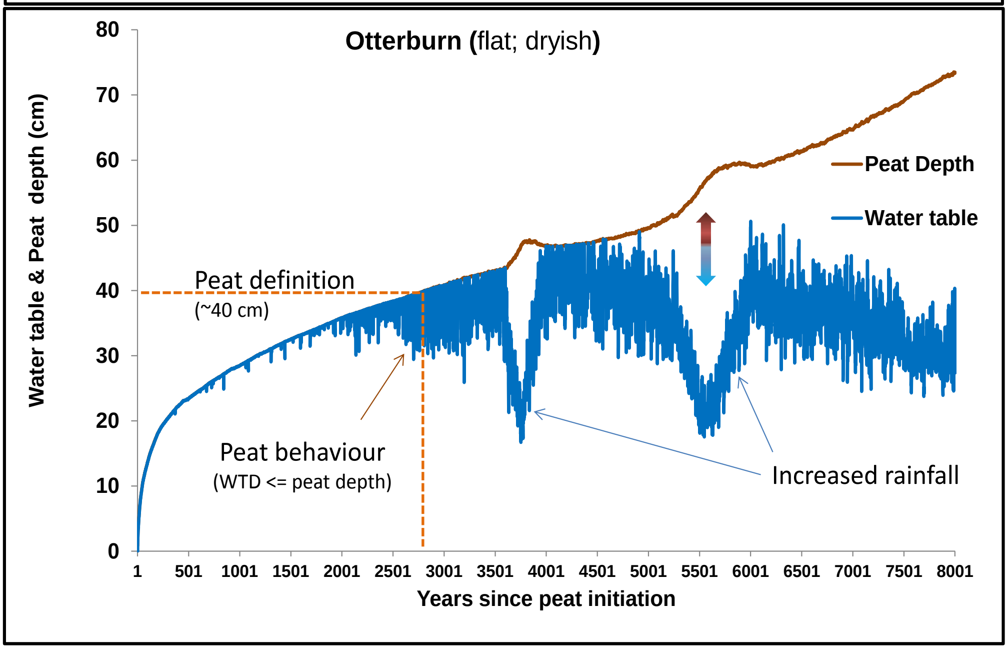


Fig. 142 MILLENNIA model (Heinemeyer et al., 2010) prediction of peat depth (brown) and water table depth (WTD; blue) at two sites with different mean annual temperature (MAT) and annual precipitation (MAP): (**top**) a typical "deep peat" at Moor House (flat slope 550 m a.s.l.; 5.2°C MAT and 2000 mm MAP), and (**bottom**) a "shallow peat" at Otterburn (flat 220 m a.s.l.; 7.7°C MAT and 895 mm MAP).

The presented model scenarios highlight the important role past management might play in explaining currently observed SOC stocks and the remaining extend of peatlands. Particularly peat cutting for fuel and agricultural cultivation were predicted to cause significant losses, which is in agreement with previous literature on historic peat cutting (Ardron, 1977) and cultivation of lowland peatlands (Evans et al., 2017b). The predictions based on the five sites in this study amounted to a loss of soil carbon of about 25 Mt km^{-2}, which equates (on an equal area basis) to slightly more than the amount of soil carbon still remaining across the entire area of Great Britain (to 1 m depth). However, the predictions should primarily be seen as a justification to obtain more evidence for this potential loss of soil carbon (and in relation to potential future C gains); there is a clear knowledge gap in both the magnitude and extent of the impacts of the various processes.

Moreover, the model predictions highlight the potential of soils to sequester more carbon if management would allow soils to return to their 'natural' trajectory of previously high SOC stocks, in many instances leading to organo-mineral soil and ultimately potential peat soil formation. Importantly, peat and corresponding SOC accumulation rates are much higher initially (when peat is formed) compared to accumulation rates on deep peat areas (see much steeper initial slope in peat depth increase in **Figure 142**). Therefore, considering the landscape scale extent, there is considerable long-term C sequestration potential of reverting to land management practices that allow soil C sequestration and peat formation. Importantly, considering this likely SOC storage *potential* would lead to different predictions of carbon accumulation compared to model assumptions based on *current* soil carbon stocks, including predictions by process level models (e.g. Parton et al., 1983) or bioclimatic envelope models (e.g. Gallego-Sala et al., 2010). Particularly, when comparing or identifying where to focus restoration efforts such long-term model scenarios might be informative as, for example, they highlight areas of former peatlands to have a much higher C sequestration potential than current peatlands. Further development of such model scenarios could become a future tool to identify suitable areas for maximising soil carbon sequestration with important impacts on associated ecosystem services such as water cycling and storage, particularly in relation to climate change mitigation potential via management related soil carbon sequestration.

In summary, the upscaling and modelling work described in Section 4.6 of the report has shown that:

- The MILLENNIA model was able to reproduce the variation in water table depth over the last century which had been reconstructed for the Moor House site.
- MILLENNIA simulations for the period 1825-1915, when there was active drainage of the Moor House site, showed that grouse moor management lowered water table depths (i.e. drier), lowered primary productivity and reduced C accumulation compared to an unmanaged scenario.
- MILLENNIA model simulations over a period of 8,000 years, incorporating effects of population pressure, fuel demand and climate linked to specific managements, such as peat cutting, burning and ploughing, suggested that historic management (since 1100 AD) has caused peatland organic carbon stores to be depleted by as much as 50%.
- This effect of historic management over several millennia was greater at low-lying sites close to population centres than at remote upland sites but highlighted a generally large landscape scale soil C sequestration potential when considering potential peat formation linked to land management changes.
- Such model scenarios also provide additional information for estimating long-term C storage under no management compared to possible management options, which otherwise is currently not available.
- Finally, the model findings highlight the potential of such scenario comparisons for unmanaged versus managed areas to help identify areas of potentially high future C gains, which ultimately might inform focusing targeted restoration measures, especially by identifying potential areas of past peatlands (i.e. reduced or lost due to peat cutting or cultivation).

4.7 *Comparative analysis of costs and emissions (towards a cost benefit analysis)*

A basic analysis was conducted on management costs and emissions, using the information acquired on person-hours, fuel spent to achieve either burning or mowing management, plus biomass consumption during burning. Data were obtained from the burning and mowing carried out in 2013 and 2015 per catchment per site. The person-hours for burning were consistently higher than those required for mowing (**Figure 143**; left); on average, 21 person-hours were required for burning and 14 person-hours for mowing. In contrast, the amount of fuel used by equipment was consistently much less for burning than for mowing (**Figure 143**; right). Whitendale required more time for both managements and more fuel for mowing than the other two sites. This reflected the larger catchment size, more remote access, and greater sub-contractor distance for this site, which was more remote than the other two sites. Both Nidderdale and Mossdale allowed easy access and at Nidderdale, the site had its own mowing equipment. The three sites provided a range of mowing management methods, with mowing being done by either the site management staff themselves (Nidderdale) or by a subcontractor located nearby (Mossdale) or further away (Whitendale). Moreover, all three sites are part of active shooting estates and represent a range of usual grouse moor management, with their own keepers doing the burning of heather, and hence no subcontractor costs.

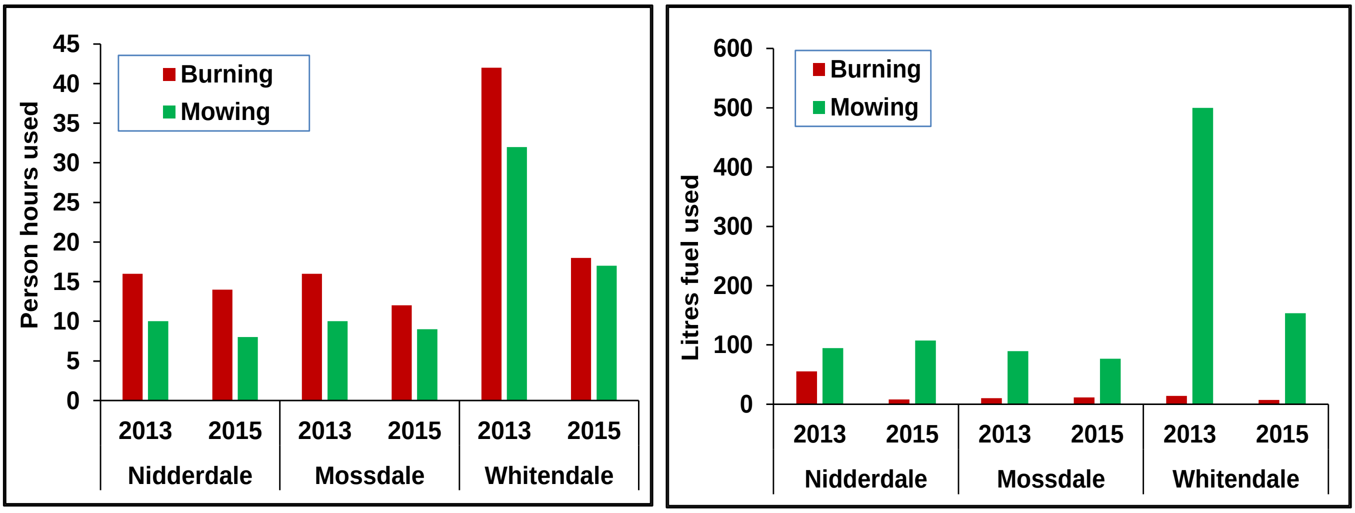


Fig. 143 The recorded (**left**) person hours and (**right**) amount of fuel spent for each management (mowing and burning) in 2013 and 2015 at the three sites.

Expressed on a per hectare basis (**Table 27**), on average burning required around 15 person hours, about twice as much as was required for mowing. Mowing required on average about 10 times the fuel amount (98 L ha^{-1}) than did burning (12 L ha^{-1}). Fuel requirements for burning and mowing per hectare were similar across sites. There were two outliers: Nidderdale in 2013 used more fuel for burning due to bad weather conditions, while Whitendale used more fuel for mowing in 2013 due the ground being very boggy, which required the use of different machinery and additional vehicle requirements (extra Argocat (ATV) and quadbike use for guidance of the tractor for mowing). The average cost for mowing (£1,227 per hectare) was more than six times that for burning (£191 ha^{-1}); these values included sub-contractor costs (for mowing at Mossdale and Whitendale only) and the cost of gamekeeper time during burning and mowing and the total fuel costs. The running costs and fuel consumption of each cost item were based on gamekeeper and sub-contractor information and estimates, and also reflected the high fuel consumption at slow speeds. The following list provides a breakdown per cost item:

1. Man hours: £10 per hour
2. Tractor usage: £15 per hour (12.8L/hr of diesel: £1/L)
3. Argocat usage: £10 per hour (2L/hr of gasoline: £1.25/L)
4. 4x4 usage: £7 per hour (5L/hr of diesel: £1/L)
5. Quadbike use: £5 per hour (1.35L/hr of gasoline: 1.25/L)

Table 27 Recorded person hours spent at each site per year per management (burning versus mowing), the amount of fuel (diesel and petrol/gasoline) used per management (based on gamekeeper and sub-contractor information) and the associated total costs of management, all based on best estimates for man hour, running costs and fuel consumption (see list in the main text above) or total sub-contractor costs (for mowing at Mossdale and Whitendale) for the actual managed area within each sub-catchment (per hectare), see **Table 28**.

		Person hours/ha		Fuel use (litres)/ha		Cost (£/ha)	
Site	**Year**	**Burning**	**Mowing**	**Burning**	**Mowing**	**Burning**	**Mowing**
Nidderdale	**2013**	8.9	5.4	31	51	167	429
	2015	10.4	5.8	6	78	142	582
Mossdale	**2013**	16.0	7.6	10	68	181	1508
	2015	14.8	7.3	14	62	180	1607
Whitendale	**2013**	21.9	14.8	7	231	264	1346
	2015	17.5	11.0	7	100	211	1891
	AVERAGE	**14.9**	**8.6**	**12**	**98**	**191**	**1227**

The emissions of CO_2 arising from all fuel use (**Table 28**), averaged over the three sites and two years, was 43 g C ha^{-1} (or 159 kg CO_2 ha^{-1}) for mowing, three times as much as the 14 g C ha^{-1} (or 51 kg CO_2 ha^{-1}) for burning (but not including emissions from the burnt biomass). Burning heather was estimated to add CO_2 emissions of 18.3 t CO_2 ha^{-1} based on a biomass loss of ~580 g C m^{-2} (see Section 4.2.5.2). A biomass consumption rate of 11.12 t ha^{-1} was derived from the fuel loads given by Smith et al. (2011) assuming their 'best estimate' scenario assumption of 80% mature heather (mean fuel load of 1.75 kg/m^2) to 20% late building phase heather (mean fuel load of 1.15 kg/m^2). The emission factor for heather biomass burning was derived from Smith et al. (2011) assuming their 'best estimate' scenario of 1:9 ratio of 'backfire' to 'headfire' burning. Together with emissions from fuel use, this resulted in burn emissions (18.4 t CO_2 ha^{-1}) which were about 115 times that from mowing (159 kg CO_2 ha^{-1}). However, mowing also causes slow release of biomass carbon from decomposition, which was not accounted for; this was assumed to be a low contribution although a higher decomposition flux across all sites was detectable in chamber fluxes on mown compared to burnt plots over the last two years after management (**Figure 86**).

Table 28 Management CO_2 emissions based on best estimates for diesel consumption per site and management (burning versus mowing) based on gamekeeper and sub-contractor information and the resulting up-scaled per hectare emissions based on the managed areas. 1 litre of petrol and diesel was assumed to weigh 0.845 kg. Fuel used by tractors was reported directly by the contractors (as litres of diesel) whereas fuel used by other vehicles was calculated from the time usage reported assuming 5 L diesel ha^{-1} for 4x4s, 2 L gasoline h^{-1} for all-terrain vehicles, and 1.35 L gasoline h^{-1} for quad bikes. In order to calculate CO_2 emissions, the fuel usage was first converted into energy equivalents (assuming 46.5 MJ/kg for butane, 35.8 MJ L^{-1} for diesel and 34.2 MJ L^{-1} for petrol/gasoline). These were then multiplied by the IPCC (2006) Tier 1 default CO_2 emission factors (butane = 0.063 kg CO_2 MJ^{-1}; diesel oil = 0.074 kg CO_2 MJ^{-1}; gasoline = 0.069 kg CO_2 MJ^{-1}) to arrive at total CO_2 emissions per management event.

		Total emissions including butan gas (kg CO2)		**Managed size (ha)**		**Total emissions (kg CO2/ha)**		**Total emissions (kg C/ha)**	
Site	**Year**	**Burning**	**Mowing**	**Burning**	**Mowing**	**Burning**	**Mowing**	**Burning**	**Mowing**
Nidderdale	2013	84.2	140.8	1.79	1.86	**47.0**	**75.5**	**12.8**	**20.6**
	2015	20.6	215.8	1.34	1.37	**15.3**	**157.1**	**4.2**	**42.8**
Mossdale	2013	56.0	179.8	1.00	1.32	**55.9**	**136.1**	**15.2**	**37.1**
	2015	52.8	164.2	0.81	1.24	**65.0**	**132.4**	**17.7**	**36.1**
Whitendale	2013	81.7	611.2	1.92	2.17	**42.6**	**281.9**	**11.6**	**76.9**
	2015	80.5	263.9	1.03	1.54	**78.2**	**171.1**	**21.3**	**46.7**
	Average	62.6	262.6	1.3	1.6	**50.7**	**159.0**	**13.8**	**43.4**

In addition to the added CO_2 emissions, burning of heather biomass visibly (see **Figure 8**) caused the emission of other air pollutant species (**Figure 144**; left). As for CO_2, emissions of each pollutant species were calculated as the product of the area burned (ha), the biomass consumption (t/ha) and the emission factors for the pollutant species concerned (as kg per tonnes burned). Emission factors for CO, CH_4 and NH_3, like those for CO_2, were then derived from Smith et al. (2011) as described above. Emission factors from Andreae and Merlet (2001) were used to estimate emissions of the remaining pollutants (SO_2, NO_X, NMVOC, PM_{10}, $PM_{2.5}$, black carbon (BC) and organic C (OC)).

However, mowing required the burning of fossil fuel, which can cause considerable air pollution from exhaust fumes, mainly from diesel engines of the tractors. These emissions (**Figure 144**; right) were calculated taking the average litres of fuel use per 1.5 ha mowing, converting this into kg diesel (assuming 0.83 kg L^{-1} density), and multiplying this by the EMEP/EEA (2016) Tier 2 emission factors for diesel use in off-road agriculture vehicles (Sector 1.A.4.c.ii) assuming the 1991 to 2001 technology range, which was based on reported age of tractors used. This showed that the additional emissions of air pollutants from the diesel used in tractors during mowing were insignificant when compared with the emissions from heather biomass burning. Emissions from the small amount of fossil fuel used during burning were trivial compared with emissions from the heather biomass burning itself and were therefore not included here.

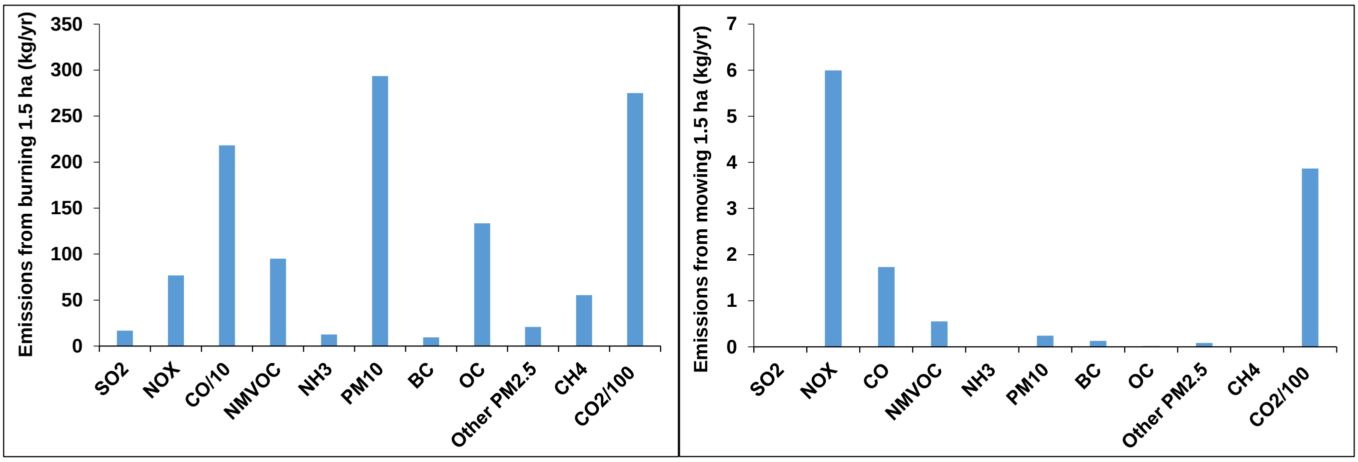


Fig. 144 Estimated emissions from (**left**) burning 1.5 ha heather moorland (i.e. the average area burnt each year within the project's ~10 ha catchments), assuming a ratio between late building (20%) and mature phase (80%) growth and the fuel loads and combustion (for further details see main text). The estimated CO_2 emissions from burning were equal to a heather biomass carbon loss of 580 g C m^{-2}. Additional pollution from tractors and other vehicles (burning diesel/petrol) during mowing is shown (**right**) as annual emissions per ~10 ha site catchment (c. 1.5 ha mown/year). Note that the scale on the y-axis is very different in the two diagrams.

After five years of only partly different management (i.e. in the rotation cycle so far only ~50% of the heather dominated catchment area has been managed differently), the three experimental sites are now in a transition period. While management costs have been recorded, the benefits in relation to effects on vegetation composition, hydrology, carbon fluxes, GHG emissions and birds are not clear; whilst some initial impacts have been found in this transition period, there were insufficient data to capture the long-term trends of any of these parameters that are essential to inform a robust cost-benefit analysis (CBA). However, robust long-term trajectories (and thus benefits) of management changes can be expected to be revealed in future years, once vegetation re-growth and a full management rotation covering the entire catchment has been achieved. This applies particularly in relation to carbon budgets, net GHG emissions, hydrology, water quality and vegetation dynamics, all of which would be central to any comprehensive CBA. Therefore, no detailed CBA is presented at this stage.

5. Concluding remarks and suggestions for future work

This project *"Restoration of blanket bog vegetation for biodiversity, carbon sequestration and water regulation"* was specifically designed to provide meaningful evidence within a practical grouse moor management context and assess: (i) possible management options to reduce heather dominance and increase 'active' bog vegetation (especially *Sphagnum* spp.), and (ii) management impacts on key ecosystem services related to carbon cycling, water budgets and biodiversity. This study is unique in providing landscape scale (catchment) replication and long-term monitoring over an initial pre- and subsequent post-management period. This project is based on a rigorous experimental setup (three paired catchments with a replicated plot experimental design comparing burning and alternative mowing managements to uncut plot comparison areas), and is designed to relate robust science to practitioner needs (e.g. on resilient bog systems and management implications for ecosystem services). All three sites are on active grouse moors with different degrees of management intensity, and they represent a range in habitat status, from less modified (Mossdale: higher species diversity and wetter peat) to more severely modified bog (Nidderdale: lower species diversity and drier peat). Therefore, the experimental set-up offers a unique long-term observatory at sites which are representative of large parts of the UK uplands in the context of grouse moor management and restoration of 'modified' blanket bog towards an 'active' state.

The UK lacks robust scientific measurements or "real numbers" for blanket bogs on carbon and water budgets, GHG emissions, and biodiversity in relation to evidence on management and climate impacts (see limited data in Smyth et al., 2015). So far, the investment in this project has delivered much to address this evidence need, but only for an initial transition period, providing a baseline against which to monitor longer-term change across a broad range of ecosystem services. The project has at least partly addressed two of the research recommendations of Natural England's Upland Evidence Review (Glaves et al., 2013, [NEER004]) for "*research on [blanket bog] restoration management, including the potential use of one-off burning and alternative treatments to reduce graminoid and heather dominance where this is an objective*" and contributed to another for "*the extension of experimental and monitoring studies of the effects of burning on vegetation and ecosystem services to a wider range of sites across the English peatland resource*".

Crucially, previous work has lacked a robust level of replication across plot-to-catchment scale, and has not covered the ecosystem services aspects jointly in a catchment-scale management context. Furthermore, no other previous work compared burning and mowing together with a pre-treatment period before management intervention to account for any site and plot differences that were present before, and thus are unrelated to management change. Most previous studies rely on a 'space for time' substitution, comparing locations with different age and management structures to infer changes through time. This is based on the assumption that there are no differences between study locations apart from those in age and type of management. This ignores the potential for differences in other important factors to occur. Notably, differences in management history between locations is very likely and could be associated with the fact that less frequently managed sites are either less accessible or wetter. Moreover, differences in slope and aspect across catchments or on plot locations can play a major role in explaining differences in soil temperature and moisture, vegetation composition and hydrology, and thus in carbon fluxes. The project also uncovered evidence that some of the findings from previous work may be methodologically flawed. For example, it was identified that: (i) claims that burning increases soil temperatures, and hence decomposition rates, might have been affected by sensor exposure to radiation; (ii) carbon losses from peat soils over recent decades that have been attributed to warmer temperatures could have been due to natural changes in peat bulk density; and (iii) studies that identified lower peat accumulation due to burning seem to have lacked accuracy in soil carbon assessment and peat depth dating.

The findings from the sub-catchment comparisons (e.g. of stream flow) clearly showed important large-scale effects of the mown management intervention that could not be captured by studies of the individual plots. Although the systems are now in a state of gradual recovery in the individual patches of the sub-catchments that

have received management (i.e. mown areas), ongoing rotational management across a sub-catchment, with an increasing proportion of the area receiving management, and the varying scale, pattern and location of patches at each heather growth stage, will affect the observed sub-catchment impacts. Moreover, at the onset of the project the monitoring plots had likely all been burned around 15-20 years ago and are still at the relatively early stages of a possible trajectory to recovery. Overall, the plot- to catchment-scale measurements highlight the importance of investigating the catchment-scale management rotation effect, and the comparison to uncut plots (ideally to include a no management catchment scale comparison), for much longer time scales. In addition, the current lack of observed differences in the cover of heather and other species for the different management treatments suggests that there is a need to consider and assess the potential of other alternative management interventions to promote recovery towards 'active' bog vegetation, e.g. through natural re-growth (layering) of very old heather within no management areas or by mowing older heather stands with high bryophyte cover, especially *Sphagnum* moss. The initial findings from the uncut/'do nothing' treatment in this study suggest that the effect of leaving such areas unmanaged over longer periods (following initial restoration treatments to, e.g., address hydrology and absence of key species, especially *Sphagnum* moss spp.) should also be a focus for further long-term study at both plot and sub-catchment scales.

The value of long-term research for both evidence-based policy (Lindenmayer et al., 2012) and scientific impact (Hughes et al., 2017) is well known. The uncertainties in this study in relation to long-term vegetation development and mean C fluxes, specifically considering mean versus median methane emissions, are a key aspect in this respect, which currently severely limit the use of the project's data for policy advice on how best to manage the land. As such, there is an urgent need for the development of a long-term network of experiments which monitor the effects of burning and other management interventions on vegetation and ecosystem services across replicated paired catchment peatland sites, as was recommended by Glaves et al. (2013). This would provide a step-change in the robustness of evidence provision and add significant value to existing research activities. For example, a recent blanket bog study (Hancock et al., 2018) only began to detect recovery of vegetation towards 'active' bog vegetation 10 years after restoration measures, and no change was detectable after 6 years. Although the overall long-term aim of this project was to assess management impacts on reducing heather cover, the 5-year period so far clearly cannot provide robust information on the long-term vegetation trajectory (i.e. vegetation is only just recovering from management and different species are now in strong competition with each other during establishment towards maturity). Long-term monitoring would also increase our process-level understanding of the environmental and ecological controls impacting the functioning (e.g. net methane emissions) of these nationally and internationally important ecosystems, and provide strategic underpinning evidence for both government policies and practical measures in relation to sustainable management and restoration of the UK's peatland resource. Such a long-term network would also greatly assist in addressing remaining knowledge gaps in relation to the development of robust UK-specific GHG inventories and emissions factors (as currently hardly any grouse moor blanket bog data are available), and in providing a sound basis for practitioner payments for ecosystem services (including flood mitigation) and carbon finance schemes such as the IUCN UK's Peatland Code.

No attempt was made to reconcile the differences between the three sites in burning and mowing methods and outcomes, as it was deemed that the between-site variation probably encompassed the range of burning and mowing techniques across Northern England. Therefore, any findings about the burnt and mown areas that emerge consistently from all three sites are likely to be directly attributable to the management itself, and are likely to be applicable to other sites across England and particularly the Pennines. However, although there was a range in site wetness and vegetation composition, all three experimental sites are modified heather-dominated blanket bog. Consequently, to interpret the effects of management intervention in relation to a transition to more natural bog systems, ideally an unmanaged and/or relatively 'pristine' catchment would need to be added to the experimental comparison, offering a 'true' control.

To capture management effects in a meaningful long-term perspective, a full management cycle over an entire catchment area needs to be monitored. For example, this longer-term monitoring would allow the identification of an optimum management rotation period based on the cumulative carbon uptake for each management scenario, i.e. a threshold age after which the carbon balance becomes close to zero and rejuvenating the vegetation would increase the long-term net carbon accumulation. It would also be able to assess if the assumption of "wetter is better" is true for the net GHG emissions of blanket bogs, as net methane emissions might outweigh any peat carbon gains; there might well be an optimum water table depth for achieving an optimum net GHG emissions which could be supported by management (yet mowing might cause too wet a bog under some conditions as seen for cranefly emergence). The higher methane emissions, and their important impact on the overall carbon and greenhouse gas balance, observed to date, especially at the wettest site, could be a result of a transition towards a more natural bog, but only longer-term monitoring will provide the crucial evidence on this. The project's stakeholder and practitioner focused approach (i.e. inclusive Project Advisory Group) has the potential to show 'win-win' scenarios based on a realistic outcomes approach (this particularly links to predictions on flooding, net carbon flux and GHG balance and bird populations) and aligns with Harper et al.'s (2018) recommendation for a prioritisation of a "collaborative effort incorporating the full range of stakeholders". Obtaining robust long-term data on both aspects (i.e. net carbon balance and net GHG emissions) would provide much needed evidence in relation to best practice management for ecosystem services in a holistic way (i.e. considering management impacts on carbon, GHG, water and biodiversity together).

The project highlighted management impacts at the wider landscape scale, such as burning causing much higher net CO_2 emissions than mowing and additional air pollution, whilst mowing reduced stream flow and thus should be considered in natural flood management mitigation measures. However, some contradictions in findings regarding the impact of burning on the net carbon balance remain. Whilst burnt plots lost considerably more carbon than mown (or uncut) plots, particularly due to large losses during biomass combustion (and not yet accounting for mowing emissions from long-term brash decomposition), the actual peat carbon accumulation at all three sites on burnt plots was similar to previous findings for unburnt plots (Garnett et al., 2000). In fact, peat cores from all three sites showed a positive correlation between peat carbon accumulation and charcoal amounts in relation to burn frequency since the 1700s (which was linked to increased bulk density and likely reduced peat decomposition rates). However, no comparison to an unmanagement core was done and peat cores were taken from relatively flat areas and most likely negative burning impacts on the carbon balance compared to benefits from mowing (with leaving brash) might be expected on slopes (i.e. erosion and related export of DOC and POC). Of more general interest, this project uncovered potential for ericoid mycorrhizal associations with heather to break down old peat carbon, potentially limiting the long-term and previously assumed recalcitrant peat carbon pools. Moreover, a peat core incubation study revealed that after an extended dry period peat shrinkage was followed by full recovery for *Sphagnum* moss dominated peat but only showed a partial recovery for *Calluna* and *Eriophorum* dominated peat. This strongly indicated the importance of *Sphagnum* moss for peat resilience against drought impacts. Finally, over much longer time scales, modelling scenarios revealed peat cutting and arable cultivation to be responsible for much larger carbon losses than from rotational burning and highlighted the considerable soil carbon mitigation potential, for large areas of land, of long-term unmanaged areas or future management towards supporting or recreating peat forming conditions, particularly on areas affected by high past peat loss.

In summary, notwithstanding the discussed limitations of the so far short-term study and the lack of an overall unmanaged catchment-scale control, the findings across the various sections of the report can be summarised in a management effect matrix (**Table 29** on the following pages) highlighting observed or likely ecological impacts and ecosystem services benefits.

Table 29 (following pages) Management effect matrix showing direction of actual change (+ increasing, - reducing, (+) or (-) minor, (NC) no change) and biodiversity/ecosystem services (ES) effect (green +ve, red -ve, with the strength of tone indicating the effect degree) in response to management mainly comparing burning to mowing (also considering the individual plot-level managements: mown with (LB) or without (BR) brash) or either management to uncut plots in relation to plot and catchment scale measurements presented in this project (i.e. the most important figure and table references are provided). Note: the interpretation needs to consider the so far limited post-management monitoring period of 4 years only, longer term impacts are likely to be different as vegetation reaches maturity (see Hancock et al., 2018) and the entire catchment management changes (see Harper et al., 2018). Sph. refers to *Sphagnum* spp., Eri. to *Eriophorum* spp., Call. to *Calluna vulgaris*, 'Bare/brash/burnt' refers to the combined cover of bare, brash or burnt ground, UER refers to the Upland Evidence Review (Glaves et al., 2013) and MftF stands for Moors for the Future.

Variable	Burning	Mowing	Notes
Vegetation composition (cover & abundance)	**Calluna +**	**Calluna +**	**More rapid increase after mow reflecting better stem regrowth, but more seedlings after burn [Fig. 33] and equally high cover after c. 3 yrs so little evidence of benefit in reducing cover initially (i.e. red; though both still at lower cover after 4 yrs [<40% c.f. c.80% pre-treatment; Fig. 34] suggesting likely 10-15 yrs recovery (similar to Hard Hill, Moor House, data for burning, UER). Limited evidence that increase is reduced on wetter sites with more Sph. which may be important [Fig. A3.2b,c, Appendix 3]. Repeat mowing might be worth testing or mowning/burning older stands with likely poorer Call. recovery or adding Sph. to very old stands or natural degeneration.**
	Cotton-grass spp. +	**Cotton-grass spp. +**	**Greater *Eri. vaginatum* [Fig. 35] cover after mow but overall already higher pre-management [Fig. 34]. Potentially beneficial re. function but over-dominance can be an issue esp. on grazed sites (increase in Sph. more important but trajectory still uncertain).**
	***Sphagnum* NC**	***Sphagnum* (+)**	**Little change overall, only at high cover on one site with evidence of an increase after mowing [Fig 37] (although 'top view' cover declined again after an initial increase in yr 4 [Fig 34a]), and in total cover overall in yr 4 [Fig. 34b] which revealed the onset of a possible different trajectory towards more Sph. cover after mowing (especially *S. capillifolium*) and lower cover after burning. Differences between sites suggest that where Sph. absent or scarce may need re-introduction following treatments otherwise risk of re-establishment of Call. dominance. Moreover, mowing possibly beneficial in spreading Sph. propagules in brash mix.**
	Other bryopytes (+)	**Other bryophytes NC**	**Decline then increase after burn (but still below starting cover) and less so after mow [Fig. 34a,b], esp. *Hypnum jutlandicum* and *Campylopus introflexus*, but overall already some pre-treatment differences and declining after 2 yrs (though this reflected 'overstorey' under-recording [Fig. 34a vs 34b] as main spp. at high cover in total cover pre-treatment and uncut [Fig. 35C]). Changes likely to be species-specific, with pioneer colonisers, especially 'acrocarps', benefiting from interventions. Other studies have shown reduced bryophyte biomass, especially 'pleurocarps', after burning (UER).**
	Bare/brash/burnt +	**Bare/brash/burnt (+)**	**Greater after burn than mow but then declined after 1st yr and similar by 4th yr [Fig. 34]. Other studies show increase in bare ground after burn (UER), which has been linked to increased DOC (UER). Bare ground was highest over the combined post-management period for brash removal plots, whilst brash/dead/burnt cover was highest on burnt plots [Fig. 35].**
Vegetation Species Richness	**NC**	**NC**	**Sites differed in species richness with the driest site showing the lowest richness [Fig. 39], which genereally increased with plot size (greater on 5x5m plots). No difference between managements during the post-management period.**
Vegetation Diversity	**NC (+)**	**NC**	**'Effective no. spp.' [Fig. 40] increased after burn but only to same level as other treaments, including uncut, from lower before. However, there was an indication of different long-term trajectories for recovery on burnt vs. mown plots [Fig 34b]. Overall diversity decreased from wettest to driest site, although the driest site showed lowest and the wettest highest total Sph. cover [Fig. 38].**
***Sphagnum* addition**	**NC**	**NC (+)**	**Added as Beadamoss in yr 2. No sign of growth till yr 4 when *Sph. capillifolium* in two mown plots on one site where absent before. Similar delayed establishment reported in other studies (e.g. MftF), so still possible to detect impact in the future.**
***Calluna* height**	**-**	**-**	**Decreased after management, initially mostly on burnt (greater regeneration from stems on mow vs. from seed on burn) but by yr 4 no difference (both ~15 cm, well below uncut ~35 cm [Fig. 33]) and both recovering becoming similar to uncut plots (thus both as red).**
***Calluna* biomass**	**L:W +, LAI - NC**	**L:W +, LAI - NC**	**Though 'leafy to woody ratio' (L:W) higher 3 yrs after burn and cut c.f. uncut and pre-teatment, Leaf Area Index (LAI), and hence quantity of leaves, much greater pre-treatment (Fig. 29), so likely similar direction of long-term recovery to pre-management levels.**
***Calluna* nutrition**	**K+, Mn (+)**	**K+, Mn (+)**	**Nutrition value increased equally on cut and burn c.f. uncut in terms of N, P, Mn, Mg & K but not other elements (Table 6), though this may only be an issue re. grouse consumption for K and possibly Mn. Even then, grouse can occur at high density on unmanaged bog (e.g. Moor House NNR) and uses other food sources.**

Variable	Burning	Mowing	Notes
Microtopography	NC	(-)	Mowing reduced microtopographic diversity [Fig. 17-18] by removing some of the hummock tops, but effect may be short-lived.
Peat pipes	NC	NC	No difference in peat pipe numbers (manual GPR surveys) between managements or c.f. uncut plots [Table 4]. Average number similar to previously reported blanket bog values.
Peat depth accumulation	NC	NC (+)	No real difference in peat accumulation (manual GPR surveys) between management or c.f. uncut plots but indication to become greater on mown than burnt plots [Table 3]. Average number within the range of previously reported values. Peat rods offer additional long-term validation. However, peat shrinkage/expansion could hide changes of about 2.5 cm [Fig. 97 & 99] or more [Fig. 101].
Peat carbon accumulation	(+)	n.a.	Peat records of dated time periods for burnt peat cores revealed higher C accumulation during periods of more frequent burns (for some periods since ~1700 AD [Fig. 93]) with higher charcoal abundance and bulk density but lower organic carbon content [Fig. 94].
Peat physical properties	NC	NC	Bulk density or peat depth did not change due to management (only a natural peat moisture [Fig. 99] and a possible charcoal impact [Fig. 94] were observed). Soil temperature averages were unaffected by management, but brash cover and uncut plots show smaller temperature ranges (reduced maxima and minima) and burnt plots showed slightly increased maximum temperatures [Fig. 70].
Peat chemical properties	NC	NC	Peat pore water pH did not change due to management or c.f. uncut plots [Fig. 52]. However, pH increased over time by one unit, which seemed to be related to a general recovery from acidification, but also partly related to a soil temperature increase of about 1 ^{o}C over 5 yrs. Peat chemistry (in the surface 5 cm) indicated surface charcoal accumulation in burnt compared to mown plots and mown plots showed increased surface lignin concentrations (when leaving brash) [Table 23].
Peat pore water quality	NC	NC	Peat pore water DOC, UV absorption spectra did not differ between managements or c.f. uncut plots. However, vegetation type did explain some of the observed variability with > SUVA under more sedge and Sph. cover and < SUVA under more Call. cover [Fig. 57]. Overall pH increased by 1 unit over 5 yrs (partly in relation to soil temperature) but without any management impact.
Flow water quality	NC	(-)	As for pore water, streams showed an increase in pH [Fig. 58] but no change in conductivity. Stream water did not show any management differences for DOC or UV absorption spectra but showed seasonal and interannual differences [Fig. 60] related to air temperature and rainfall. Only stream P concentrations increased significantly in mown catchments after management [Fig. 63].
Water table & soil moisture	(-)	(+)	Water table depth [Fig. 46-47] and surface soil moisture [Fig. 115-116] and soil moisture under comparable water tables [Fig. 130] was lower on burnt than on mown plots after management but with considerable site differences and seasonal fluctuations.
Stream flow	+	-	Stream flow was reduced after mowing (higher flow loss on burnt), which increased proportionally with an increase in catchment management area [Fig. 11 & Table 12]. Overall, mowing was predicted to result in reduced peak flooding downstream [Fig. 50].

Variable	Burning	Mowing	Notes
Cranefly emergence	-	(+)	Indicated lower emergence on very dry areas [Fig. 114] (and lower soil moisture vs. water table depth relationships on burnt plots [Fig. 130]) increased cranefly emergence on wetter (mown) areas in the dry years but indicated possible reductions in wetter years [Fig. 118-119], particularly on already wet sites. Emergence traps indicated an optimum soil moisture range of between 80%-95% [Fig. 114].
Cranefly abundance	(-)	(+)	Overall greater abundance in mown than burnt catchment transects [Fig. 122 & 124] but with the opposite impact on emergence in plots on the wettest site.
Bird populations (i.e. three cranefly dependant upland spp.)	(-)	(+)	Models predicted a decline in populations for all three bird species (golden plover, dunlin and red grouse) under predicted warmer and drier future (2050-2080) summer scenarios [134-136], with greater reductions under burn than mown management (particularly when leaving brash). However, modelling did not consider the possibility that on very wet sites mowing could result in water logging causing death of cranefly larvae and thus reduced bird numbers (as was indicated by cranefly emergence in relation to soil moisture). Golden plover predictions including vegetation height, which was lower overall on managed plots (no significant difference between burnt vs mown) vs. uncut catchment areas, showed initial post-management benefits of either management [Fig. 125].
Soil respiration (CO_2)	(-)	(+)	Indication of slightly reduced soil respiration and temperature sensitivity (Q_{10}) on burnt compared to mown (with brash left) plots [Table 14], particularly the top 5 cm peat layer (which respired a very high proportion (~60%) of the top 20 cm) [Table 22].
Net ecosystem exchange (CO_2)	NC	NC	Initially (2013) higher net C losses after management on burnt vs. mown plots, but 4 yrs after management no difference with both mamagements overall loosing C [Fig. 86] compared to net gains by uncut plots with highest 5 yr mean C gains on the wettest and a small loss on the driest site [Tabel 15].
Net methane emissions (CH_4)	(-)	NC	No difference between burnt and mown plots for net methane emissions (although uncut plots showed higher fluxes than re-grown burnt plots) [Tabel 16]. Very high site and interannual variability in net emissions and particularly in mean vs. median flux estimates; overall highest flux on the wettest site with significantly postive relationships with water table, soil temperature and sedge cover. Therefore, emissions likely to become lower on burnt (c.f. mown) plots due to lower water tables and less *Eri. vaginatum* cover.
Net Ecosystem Carbon Balance (NECB)	NC but (+ i.e. loss) when incl. biomass	NC but also (+) when incl. brash decomposition	Overall uncut areas a small net C sink [Table 18], but this depended on using the median vs. mean of C fluxes - particulalry methane fluxes, which were highest in the last two post-management yrs. when both managements showed about 8 times larger C losses than uncut areas [Tabel 19], mainly due to very low net ecosystem exchange (net C uptake). However, burnt areas showed the greatest C loss when including emissions from burnt heather [Fig. 144] (dark red) although C loss for mown (brash decomposition) also likely to increase.
Net Greenhouse Gas Emissions (net GHGs) in CO_2 equivalents)	NC but + (i.e. loss) when incl. biomass	NC but (+) when incl. brash decomposition	Uncut plots had lowest net GHGs with no difference between burnt and mown plots but greatest net GHGs on managed sites overall (but particularly on the burnt plots) for the wettest and lowest on the driest site [Tabel 21]. Site and interannual variability very high and overall high changes in GHGs depending on using the mean or median for net methane emissions [Tabel 20]. But as for NECB, GHGs were greatest on burnt plots when including the emissions from biomass burning (red) but mown did not include brash C losses.
Air quality (pollution)	+	(+)	Larger amount of air pollution (e.g. SO_2; NO_x; particulate matter) from burning than mowing equal areas, mainly due to burning heather (i.e. biomass) [Fig. 144], with mowing resulting in overall much lower emissions and pollution from vehicle use (i.e. diesel) [Fig. 144].

Importantly, as the effects of a complete change in management practice and vegetation re-growth require time, particularly in cold, wet and thus slow-growing upland ecosystems (as shown by Hancock et al., 2018), the project was conceived as a long-term study with an initial period running from 2012-2017, albeit with no guarantee of funding beyond that. The findings in this report, as evidenced in the above impact matrix (**Table 29**), strongly support extending the initial 5-year period, as it revealed major site differences and management impacts, some of which are still increasing (e.g. water), some of which are just emerging (e.g. vegetation), some of which are disappearing (e.g. carbon) and for some of which there are conflicting findings to date (e.g. craneflies).

Whilst this project focused on particular issues around key ecosystem services it also highlighted potentially far reaching impacts beyond the upland landscape (e.g. flooding) and important interlinkages between investigated effects (e.g. soil chemistry and soil microbes) as well as potentially wider implications (e.g. livelihoods). The below schematics (**Figure 145**) outline the key processes in relation to key ecosystem services with linkages across the landscape scale as impacted by management; the schematic highlights that in addition to the direct impacts measured in this project (**Figure 145a**), any overall assessment of upland management change impacts might also need to consider the implications for health/well-being, recreation/trourism and livelihoods/economy including impacts on farming and drinking water (**Figure 145b**) across the wider landscape and economic area.

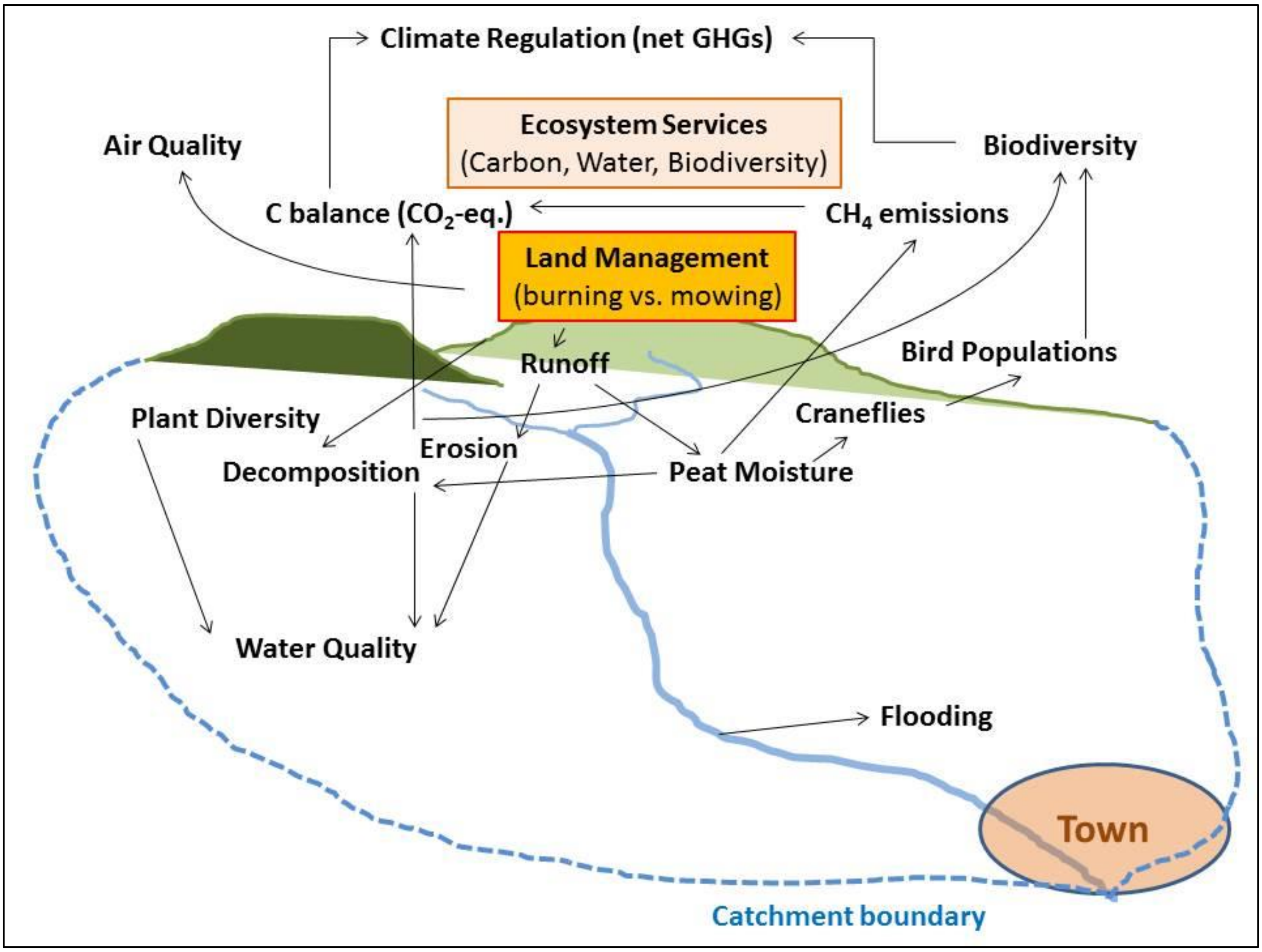


Fig. 145a Schematic network of the main linkages between management impacts on plant-soil-water processes and associated key ecosystem services aspects (related to carbon, CO_2-equivalents (CO_2-eq), greenhouse gas emissions (GHG) water and biodiversity) across a generic catchment area containing upland areas with blanket bog. Processes and parameters investigated in this project are shown in bold.

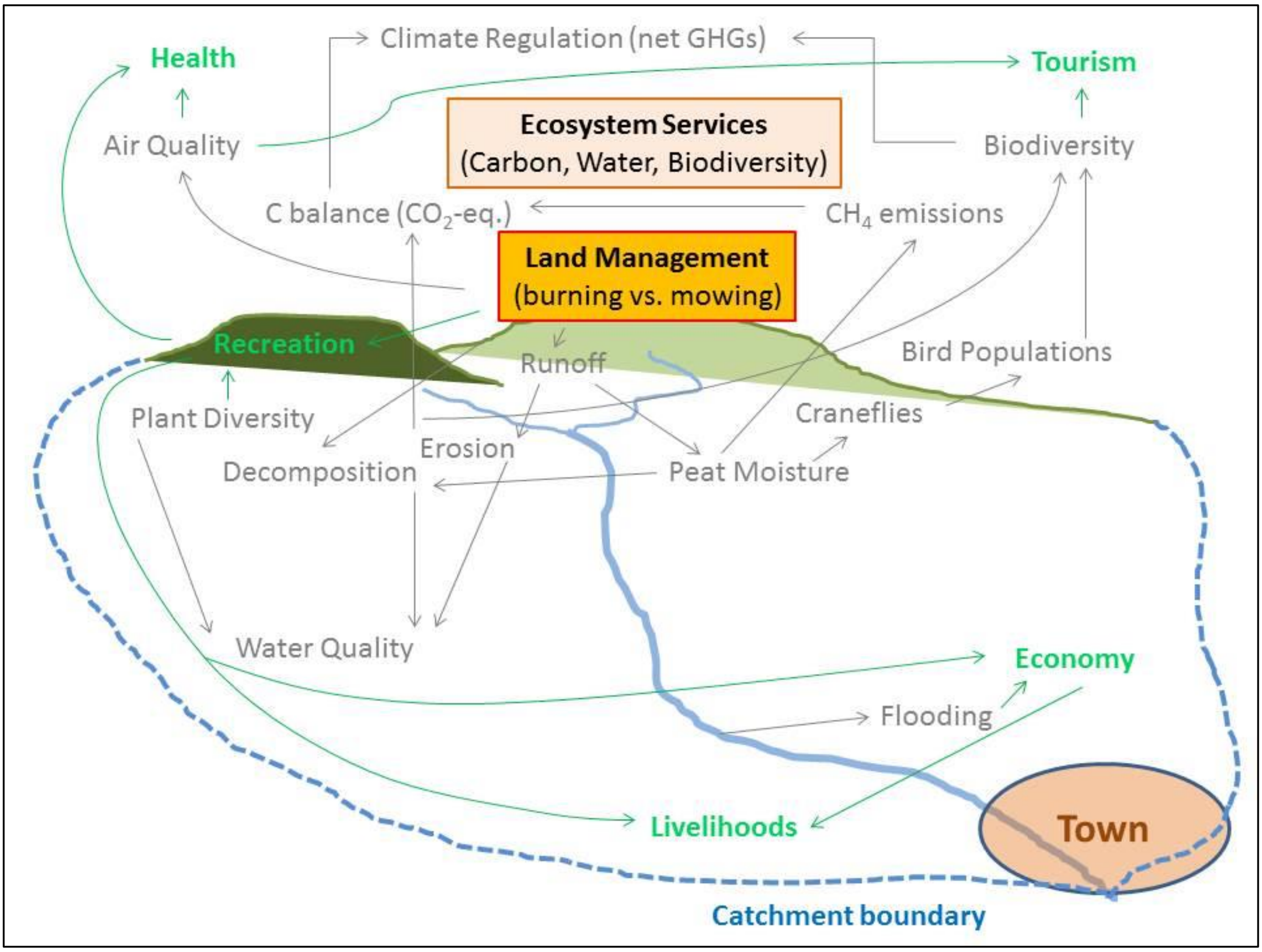


Fig. 145b Schematic network of the main linkages between management impacts and key ecosystem services across a generic catchment area (as Fig. 145a). In addition to processes and parameters investigated in this project (shown in grey) additional impacts on key ecosystem servises are also indicated (green). Note that these impacts are reaching far beyond the upland area with indirect effects on livelihoods in cities connected by water (e.g. drinking water), food (e.g. soil health) and economy (e.g. flood impacts).

In addition to the need for further monitoring, specific research needs have been identified as a result of the current project. Whilst these research needs could in principle be assessed anywhere, future work would clearly benefit from being conducted within the present project context and available data. We identify several areas in need of further research in relation to the original objectives to support policy through (i) identifying the effectiveness of alternative management methods in reducing heather dominance and encouraging 'active' bog vegetation and (ii) providing evidence on management impacts on ecosystem services, ideally also incorporating a full catchment-scale no management scenario:

Vegetation cover assessment so far provided only a limited insight into management impacts in relation to reducing heather dominance and encouraging 'active' bog vegetation. The range of management interventions assessed could be expanded. Vegetation recovery after repeated mowing could be experimentally trialled on the existing plots and compared to the paired one-off mown plots, as could mowing of additional plots of different heather ages, and re-growth by natural layering of very old and degenerated heather stands. The gradual natural opening up and natural layering of the degenerate or 'rank' heather cover could, even without management, result in lower heather cover and more diverse vegetation including more *Sphagnum* and other moss species within the protected yet open areas. Ideally this would involve an assessment of the actual heather age (i.e. counting of growth rings) and could also include additional *Sphagnum* application trials (e.g. plugs) and a no management comparison. Finally, heather control using herbicides could be assessed (e.g. within a controlled peat core incubation study).

Water quality (carbon and chemistry) showed no clear differences between management at the catchment level (i.e. stream flow), despite plot-level effects in relation to vegetation composition and indications of increased phosphorus export, probably from decomposing brash, in the mown treatment. This mechanism may also influence N export, and measurements are needed to test this, specifically in relation to potential eutrophication of streams and upland reservoirs. There were indications of impacts on water quality parameters (i.e. DOC, UV spectra and SUVA) that are specific to particular types of vegetation and of soil microbial community impacts on carbon cycling (e.g. through mycorrhizal associations). More detailed field and laboratory incubation studies are needed specifically to assess the linkage of vegetation types to soil microbial communities affecting carbon input, decomposition processes and carbon compounds, and hence provide a clearer link to vegetation mediated changes caused by management intervention.

Net ecosystem exchange and **decomposition** fluxes in relation to catchment scale **carbon budgets** were only assessed on relatively flat and heather dominated areas. Particularly slope and non-heather areas (e.g. rushes) are so far uncertain factors, specifically in relation to water tables, soil moisture and erosion, requiring expanded monitoring across the catchment and specifically considering eco-hydrological processes.

Methane emissions increased over the last two years, significantly altering the C balance and the **net GHG emissions** for all sites, but particularly at the wettest and less modified site. Since it is unsure if these are anomalous years or part of a long-term trend, this severely limits the extrapolation and interpretation of management and site effects on both parameters. The net GHG emissions require assessing the impacts of natural variation in the field, in addition to that of further controlled experimental laboratory studies in relation to vegetation composition and a possible mean water table depth threshold for limiting net methane emissions. In particular, there is a need to assess the reasons for the unusually high methane fluxes found in 2015 and 2016, particularly at the wettest site, and the implications of the predicted "future climatic norm" of wetter and warmer winters.

Peat accumulation rates can only be captured accurately in the long-term; therefore, the installed fixed datum peat rods offer a unique opportunity to relate site- and management- specific carbon fluxes to carbon stock estimates and long-term monitoring. A more detailed **peat depth/age profile** assessment is required (ideally including other sites) linking past management over well-defined time periods to carbon accumulation rates, as well as revealing past peat losses due to fuel extraction, in order to avoid misinterpretation of peat accumulation rates over time and linking historic impacts to currently observed C stocks and fluxes. Such an assessment together with a SOC model framework would also inform targeting restoration work towards areas of maximum C gains (i.e. based on identifying potential areas of high past SOC losses and suitable conditions for peat formation). Moreover, the interpretation of differences in recent C accumulation in terms of burn frequency requires information on past vegetation composition (e.g. plant macrofossils and pollen records).

Peat chemical composition is a critical factor influencing decomposition processes and carbon cycling. Therefore, it needs further assessment in relation to climate and management scenarios with a wider range of chemical analyses which are relevant to carbon compound composition, and hence the rates of decomposition processes. In particular, there is a need to characterise recalcitrant components, and the role of soil microbes in their degradation, to determine impacts on carbon cycling in blanket bogs. The observed **charcoal impact** on peat physical structure (bulk density) and hence hydrological conditions (through runoff and peat moisture), as well as the potential impacts on soil microbial communities and decomposition processes, require further assessment as they could explain the unexpectedly high carbon sequestration rates under burning identified by one of two C budget/accumulation methods. Such aspects relating to C-cycle processes and eco-hydrological feedbacks also need to be considered in future model developments and scenarios.

Cranefly emergence was increased by mowing in the first dry year after management but was decreased by mowing in the following two wet years, possibly indicating an upper moisture limit of around 95% for cranefly larvae survival (possibly due to drowning of larvae). Data for more climatically contrasting years or laboratory incubation studies are needed to be confident about this interpretation, and hence to predict possible negative mowing effects on cranefly abundance during wet seasons, years or on generally wet sites.

Bird impacts of different managements could so far only be modelled based on one drier year, limiting the assessment of the potential long-term benefits of mowing on soil moisture, cranefly numbers, and thus bird breeding and survival. **Bird models** so far do not specifically include impacts of vegetation structure on red grouse populations; the main question remains how much heather, and of which age structure, can provide the shelter and nutrition to support adequate grouse populations. Moreover, possible negative mowing impacts on birds via moisture (drowning) impacts on cranefly abundance and micro-topographic impact (hummocks) should be considered further.

In conclusion, the research so far has delivered much needed robust evidence over a substantial but only initial monitoring period, effectively providing a short-term baseline, together with an initial, short pre-treatment (no management change; BACI) period against which to monitor meaningful long-term change across a broad range of ecosystem services (carbon, water and biodiversity). Importantly, previous work has lacked this robustness of the experimental approach with a plot- to catchment-scale replication as part of a BACI design and did not cover these ecosystem services aspects jointly in an alternative management context.

Carbon and water cycling are greatly affected by a combination of site, climate and management. To tease apart site and climate effects from management impacts requires long-term datasets, yet there are no other long-term data for blanket bogs under active grouse moor management. A recent meta-analysis study by Harper et al. (2018) highlighted the relatively low number of studies on burn management impacts on key ecosystem services (i.e. water storage, water quality, carbon storage), the remaining overall uncertainties (particularly the issue of charcoal and long-term carbon storage) and the need to assess alternative management (specifically cutting). Therefore, the current work, whilst providing much needed policy evidence advice on burning and alternative blanket bog management impacts on ecosystem services, also has the potential to demonstrate realistic practitioner-relevant outcomes with multiple benefits to all stakeholders. Notably, Harper et al. (2018) emphasise the need for long-term monitoring in relation to providing robust evidence ("*it is crucial that research includes data from the full post-burn recovery period*") as part of supporting a well-informed and unbiased debate ("*a collaborative effort incorporating the full range of stakeholders must be prioritised*") on management impacts on ecosystem services and, importantly, recommend prioritising monitoring over more than one burn rotation. Sites are now in a transition phase, with past burn (and possibly other management) history still affecting catchment areas and soil processes; to capture management effects in a meaningful long-term perspective, it is of utmost importance to monitor at least a full management cycle over the entire catchment area, allow re-growth of mature vegetation and ideally include the anticipated repeated mowing management as a possible tool to reduce heather dominance and support a transition towards 'active' bog vegetation.

Finally, most of the previous studies finding significant impacts of management (e.g. on water quality) compared burning with so called 'no-burning' rather than with mowing as in this study, or in the few cases that have done so, there has not been adequate replication nor a BACI approach. Thus if both burning and mowing showed effects in this study, e.g. on DOC, the difference between the managements may be less compared to a no management intervention (i.e. uncut), which was only tested in the experimental setup at the plot-level and not the sub-catchment scale. Moreover, large 'no-burn' areas in the now mown sub-catchments have all been burnt in the past and have thus also been modified compared to an unmodified bog. Thus, what is of most interest is how both burning and mowing (and specifically their vegetation, hydrological and associated ecosystem services trajectories) compare with unmodified or least-modified and ideally, near natural bog systems. Such studies should ideally also include assessing possible natural heather cover reductions in the long-term via re-growth from degenerated heather. Furthermore, specific experimental research and monitoring is needed to assess long-term carbon budgets and GHG emissions in those habitats and if there is a water table threshold for achieving an optimum outcome for both. Clearly including 'pristine' bog sites in future monitoring would allow an assessment of what impact the management interventions have on the restoration trajectory towards unmodified or less modified bog, i.e. are both management practices potential 'blockers', which in effect are repeatedly resetting the ecological trajectory back compared to no intervention? Notably, Harper et al. (2018) conclude that "*prescribed burning, under a changing climate, could either be a useful land management tool or a highly damaging process if implemented without sufficient impact research. Based on the current knowledge it is still unclear which category prescribed burning falls into in the UK*". The current report supports this view and highlights the need to include other alternative managements (specifically repeated mowing, herbicide treatment and no management with potential for additional *Sphagnum* (as plugs) addition, also on burnt plots) in an overall holistic ecosystem services impact assessment.

6. Knowledge transfer and dissemination

We have regularly participated in knowledge exchange. This has taken place at the start of the project (workshop in 2012; Annex A.1), during the project (e.g. through annual reports to the Heather Trust, conferences and site visits), and at the end (e.g. workshops in 2017). It has also taken place throughout the study via the project's website (http://peatland-es-uk.york.ac.uk/). Knowledge exchange has involved the scientific community, government agencies and other stakeholders such as land-users (e.g. gamekeepers and farmers), managers (e.g. shooting associations), owners (e.g. United Utilities, Yorkshire Water) and land user groups (e.g. Moorland Association). Representatives from these groups were part of the Project Advisory Group (PAG) which met annually and discussed progress (annual progress reports), planned future work and suggested additional work. Moreover, we circulated shorter progress reports to a wider group of interested stakeholders, which was selected by the PAG.

So far, four scientific papers have been published from the work (Carroll et al., 2015; Heinemeyer & Swindles, 2018; Heinemeyer et al., 2018; Morton & Heinemeyer, 2018) and several more are in preparation and will be submitted to international journals. However, most of the results have already been presented at several national and international conferences (including invited keynote lectures) during 2012 – 2017; these included meetings of: the British Ecological Society (BES); the British Soil Science Society (BSSS); the IUCN UK peatlands; UKEconet; Soil Organic Matter (SOM) and Biogeomon. Moreover, summaries of the science findings were presented to a special peatlands working group at the United Nations Framework Convention on Climate Change (UNFCCC) and a general project update was presented to Natural England staff, and other upland interest groups, via a webinar in 2017. Further knowledge transfer was undertaken by participating in several workshops including one on Payment for Ecosystem Services and several presentations to upland user groups including the Upland Hydrology group and the Yorkshire Peat Partnership (YPP). Finally, the project was presented at site visits at all three sites with representatives from the local farming community, upland user groups (e.g. Areas of Outstanding Natural Beauty (AONB), Moorland Association), water companies and government agencies such as Natural England (NE) staff, including its Chief Scientific Advisor, Tim Hill and the Chief Executive, James Cross.

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