**Experimental evidence for sustained carbon sequestration in fire-managed, peat moorlands**

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**Peat moorlands are important habitats and in the boreal region, where they store ca. 30% of the global soil C. Prescribed burning on peat is a very contentious management strategy widely-linked with loss of carbon. Here, we quantify the effects of prescribed burning for lightly-managed boreal moorlands and show the impacts on peat and C accumulation rates are not as bad as is widely thought. We used stratigraphical techniques within an unique replicated, ecological experiment with known burn frequencies to quantify peat and C accumulation rates (0 managed burns since ca. 1923, 1-burn, 3-burns, 6-burns). Accumulation rates were typical of moorlands elsewhere, and were only reduced significantly in the 6-burn treatment. However, impacts intensified gradually with burn frequency; each additional burn reduced the accumulation rates by 4.9 g m$^{-2}$ yr$^{-1}$ (peat) and 1.9 g C cm$^{-2}$ yr$^{-1}$ but not preventing accumulation. Species diversity and the abundance of peat-forming species also increased with burn frequency. Our data challenge widely-held perceptions that a move to zero burning is essential for peat**

**growth, and show that appropriate prescribed burning can both mitigate wildfire risk in a warmer world and produce relatively fast peat growth and sustained C sequestration.**

Peatlands are important habitats in many parts of the world covering ca. 3.8 x $10^6$ $km^2$, concentrated in the boreal region[1], storing about 30% of the global soil C[2], estimated at 500±100 Gt of C[3]. Peatlands occur where organic matter decomposition is prevented by low temperature and high rainfall[4]. As they are composed of dead plant material they are flammable[5], and under suitable conditions, are susceptible to fire and particularly wildfire. Fire is a natural phenomenon in many boreal areas[6] where large areas (0.03-0.24 x $10^6$ $km^2$ $yr^{-1}$) are burned annually[7-9], releasing an estimated 106-209 Tg C $yr^{-1}$, which has important repercussions for the global C cycle[3]**.** In many peatlands the natural fire return interval varies considerably from 75-425[10] to between 400 -1790 years[11]**,** but, in some regions for example the Alaskan interior, there have been recent increases in wildfire of 2.4% per year between 1943-2012[6]. As prescribed fire is often used to suppress wildfire[6,12-13], so better understanding of the relative risks and impacts of prescribed fire and wildfire is of global interest.

In many parts of the world, peatlands are left unmanaged, but large areas are also managed lightly through grazing and prescribed burning. In Norway, for example, prescribed fire has been shown to be a key part of heathland management for at least 6,000 years[14], which has produced a fire-adapted flora[15]. In the second half of the twentieth-century fire exclusion policies have been adopted in many places in western and Baltic Europe, and there have been calls to reinstate traditional burning practices to restore the functional role of fire in these areas[16]. In Canada, its use is advocated for both enhancing forest understorey diversity and forest productivity[10]. In the UK, use of prescribed burning is very contentious with heated debate on its use for moorland vegetation on peat[17-19]as it is widely-linked to ecosystem degradation, loss of C and negative impacts on water quality[18-23]. Much of the concern over prescribed burning on peat is a belief that this practice changes the vegetation type and prevents peat formation; e.g. in the UK a shift from plant communities dominated by cotton-grass *Eriophorum*/Sphagnum to one dominated by the shrub *Calluna vulgaris*. However, where prescribed burning is not used the build-up of shrubs and trees can provide a large, fire-prone fuel load which puts the peatland at greater risk from wildfire[11-13]. Wildfires can be much more damaging than prescribed fires[22-23]. Moorland managers are therefore damned if s(he) burns and damned if s(he) does not. There is, therefore, an urgent need for quantitative evidence about the use of prescribed burning on peat growth rates. Here, we quantify peat and C accumulation rates within an experiment with a known managed burning history

**Peat, a recent historic record**

Peat is a vertically-growing structure, increasing in thickness with time and laying down a stratigraphy that preserves evidence of change in local and regional vegetation[4,24], fire frequency (charcoal)[24-25], hydroclimate[26] and C accumulation[27]. Usually, these sub-fossil records are interrogated over long-time scales (1,000 to 10,000 years). However, the generation of relatively accurate age-depth profiles in peat over the last 150 years[28] has been made possible by linking stratigraphical records of atmospheric pollutant deposition[28] (stable Pb, $^{214}$Am, $^{137}$Cs and Spherical Carbonaceous Particles) calibrated against absolute geochronologies derived from radiometric dating techniques ($^{210}$Pb). Here, we have applied this integrative approach to create age-depth profiles for peat sequences within the unique, long-term, manipulative, experiment at Moor House National Nature Reserve in the north of England. This experiment is set up on a *C. vulgaris*-dominated, ombrotrophic (rain-fed) peatland. We tested one of the major assumptions underlying studies on the effect of prescribed burning on peat and C accumulation patterns: that burning or burning frequency prevents or reduces peat and C accumulation. Multiple, shallow peat profiles (n=32; <0.5m depth) were sampled in four different managed burn treatments (of 0, 1, 3 and 6 burns since ca. 1923[29]), each replicated in four blocks (Supplementary Fig. S1). Two additional master peat profiles were collected to determine chronological markers and age-depth profiles using the atmospheric stable Pb down-core record (measured by X-ray Florescence, XRF). Within these master cores, independent age control was secured by $^{210}$Pb, $^{137}$Cs and $^{241}$Am analysis using direct gamma assay producing $^{210}$Pb chronologies corroborated in part by radionuclide fall-out ($^{137}$Cs and $^{241}$Am) markers[30] for 1963 and 1986. Our age-depth models (Supplementary Fig. S3) have chronological uncertainties of ±1-5 yr (1980–2014) and ±5-13 yr (1900-1970)[28]**.** Atmospheric stable Pb (Extended Data Fig. 2) profiles were then measured for the 32 cores by XRF. The two reliable atmospheric pollutant Pb markers at ~ 1876 and 1963 were discerned in all 32-peat profiles and used to calculate dry peat and C mass accumulation rates for each profile for the two periods within the age-depth profile (1876-1963 and 1963-2016). The measured peat accumulation rates are net ones, integrating the effects of damage to the peat and subsequent regrowth

**Impact of increasing burning frequency on peat and C accumulation**

The measured results of mass and C accumulation rates (1963-2016) for the 0-burn treatment were 124.4 ± 8.04 g peat $m^{-2}$ $yr^{-1}$and 48 ± 3.3 g C $m^{-2}$ $yr^{-1}$ respectively. The C accumulation rates are in the same order of magnitude as reported literature values; 24.1 g C $m^{-2}$ $yr^{-1}$ as a long-term average for northern peatlands, and between 18 and 206.2 g C $m^{-2}$ $yr^{-1}$ from a range of UK peatlands sites[31-36]. Moreover, our values are very close to the average predicted value of 56 g C $m^{-2}$ yr (range (20 –91) derived from the entire catchment in which the Moor House managed burn experiment is situated[37]. Our measurements for 1963-2016 were lower than those from the earlier 1876-1963 period (142.1±16.1 g peat $cm^2$ $yr^{-1}$; 55.0±6.2 g C $m^{-2}$ $yr^{-1)}$ but this difference was not statistically significant (peat, t=0.97, P=0.38; C, t=0.99, P=0.37, df=3).

Prescribed burning only caused significant reductions in peat and C accumulation rates (Fig. 1a; peat $F_{3,9}$ = 5.5,0 P=0.026; C $F_{3,9}$ = 4.51, P=0.034) at the extremes between the 0-burn and 6-burn treatments; (Tukey HSD, Mass = P<0.020; C = P<0.027). As we did not detect a significant difference in vertical peat growth between burning treatments (mean 0.158 ± 0.005 cm $yr^{-2}$, n=32, range =0.116-0.202), the observed changes in peat mass must reflect a changing peat density. The different burning treatments reflect an increasing number of burns, which can be described by a linear relationship (P<0.01, Fig. 1b), essentially for each additional burn the accumulation rates were reduced by 4.9 g $m^{-2}$ $yr^{-1}$ for peat and 1.9 g $m^{-2}$ $yr^{-1}$ for C.

The burning treatments have also produced changes in biodiversity (Fig. 2). Overall diversity (Shannon-Weiner Index) increased in the 3-burn and 6-burn treatment but declined in the 1-burn one. *C. vulgaris* had greatest abundance in the 1- and 3-burn treatments and lowest in 6-burn treatment, although all increased in abundance through time. *Sphagnum* showed no significant change in 1-burn treatment but significantly increased in the 3- and 6-burn treatments, with the 6-burn one having a greater overall abundance. *Eriophorum vaginatum* showed no temporal trend but its abundance increased with increasing burning frequency.

These results debunk a number of widely-held beliefs in peatland conservation (Fig. 3). First, the belief that prescribed burning prevents peat and C accumulation was not supported because even after six burns, peat and C were both accumulating; the accumulation rates were reduced, but not stopped. We should, however, not be complacent and further monitoring is needed to better understand longer-term impacts. Second, in broad terms it is usually believed that *C. vulgaris*-dominated communities will have little peat accumulation whereas those dominated by *E. vaginatum* and *Sphagnum* will be good peat accumulators[18]. Here, the opposite was found; the vegetation in the 1-burn (and indeed the 0-burn reference plots) had the greatest accumulation rates yet were dominated by *C. vulgaris* and the plots burned most frequently with the lowest peat and C accumulation rates were dominated by *E. vaginatum* and had greatest *Sphagnum* abundance (Fig. 2)[38-39]. Taken together, these results do not support the simplistic ideas about peat accumulation and plant community type, and confirm that reasonable peat formation (0-burn treatment = 48 g C $m^{-2}$ $yr^{-1}$) can occur under a *C. vulgaris*-dominated community with lower rates under *E. vaginatum* and *Sphagnum* (6-burn treatment = 36 g C $m^{-2}$ $yr^{-1}$). It is possible that the presence of the peat-producers (*Sphagnum* and *E. vaginatum*) counter-balance the effects of more frequent, prescribed fires.

**Management implications**

At face value, these results imply that prescribed burning on moorlands should be limited in order to enhance C accumulation rates and support C storage as an ecosystem service[17-19]. Alas, it is not quite so simple (Fig. 3). Peatland conservation and its associated ecosystem services cannot be separated from potential wildfire occurrence, common in upland parts of the UK and elsewhere in the boreal region[2-3,6-11]. Wildfire is expected to be a greater problem with the drier summers predicted as the climate changes[19,40-41]. *C. vulgaris*, the dominant and increasingly dominant species in the 0-burn treatment, is a species with traits that respond positively to fire; igniting easily especially where there is a large proportion of dead material[5], as is the case in old-growth stands, regenerating quickly after prescribed burning[42] with seed germination enhanced by smoke[43]. However, under wildfire the entire plant can be killed and surface peat damaged severely [direct damage and C loss][22], and loss of bryophyte regeneration potential[44]. Thus, where *C. vulgaris* dominates over large areas, as here in the 0- and 1-burn treatments, the vegetation must be susceptible to spring and summer wildfires; previous wildfires have seen large areas damaged, loss of surface vegetation hence loss of biotic control[45], with subsequent erosion of peat by heavy rainfall [indirect damage, but up to 1m depth can be lost][46]. In such a wildfire, C losses could swamp any improvement in C accumulation occurring through a reduction in prescribed burning, especially if the peat burns. To estimate potential damage we estimated the total C concentration in the surface vegetation (820 g C $m^{-2}$) plus the amount in the surface 1 cm and 5 cm depth layers (240 and 1274 g $cm^{-2}$ respectively, Fig. 3). If these surface vegetation/peat layers were destroyed by wildfire we estimate it would take and 58 years to recover this lost C and attain the status quo. These estimates have large uncertainties (95% CL = 22-38 and 48-71 years for 1 cm and 5 cm peat loss respectively and an optimistic scenario of an immediate ecosystem recovery and a C accumulation rate of 36 g C $m^{-2}$ $yr^{-2}$ (6-burn value). Clearly, if accumulation rates were further reduced by wildfire, or if there was an extended lag-effect[11] then these estimates would increase.

Managers must consider, therefore, both the impacts of prescribed burning relative to wildfire risk in developing moorland conservation policies[47]. We suggest that for this moorland under current climatic conditions (Fig. 3) the 3-burn treatment (equating to a burn every 20 years, with some areas left unburned) would be a pragmatic solution. This approach would minimize damage to peat and C accumulation rates, maintain a mixed-moorland community with maximum diversity, and a reduced fuel-load providing some degree of resilience to wildfire. With different patches burned annually, a mosaic of stages ranging from post-burn through to old stages would be created across the landscape. These findings have implications for managed and unmanaged peatlands globally where prescribed burning is a widely-used management strategy[9,10,16]. Indeed, for northern Europe it has been argued that the recent reduction in the use of prescribe burning needs to be reversed[16]. If global warming introduces a much shorter return cycle to wildfires, then prescribed fires could be one way of reducing the damage. The unique long-term ecological experiment at Moor House National Nature Reserve shows that C sequestration and biodiversity in the fire-managed NW European boreal peat moorlands is not as bad as previously thought. The threshold burn cycle to optimise C sequestration and promote greater biodiversity may need to be shortened in areas with faster vegetation growth rates[12,47], or lengthened in peatlands with slower growth, and particularly where arboreal communities are part of the ecosystem[23]. However, our general stratigraphical approach offers a mechanism in modified form for identifying the optimal managed-burn frequencies for other locations should changing wildfire regime require a more active management strategy. The major conclusion is that prescribed burning on peatlands is not necessarily damaging. Where there is evidence of the traditions use of fire on peatlands, appropriate frequencies need to be derived, and even where there is no current management, prescribed burning could perhaps be considered for wildfire prevention in the future, especially with the projected global increase in frequency wildfire[48,49].

**Online Content** Methods, including statements of data availability are available at Nature.website.

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**Author Contributions**

RHM and RCC planned and carried out the field sampling with RR, E-LM, RL and KH. RCC led the geochemistry/stratigraphy with E-LM and RL; PA and GP were responsible for the radiometric dating; the vegetation survey and analyses were planned and performed by JA, KAA, HL, GM, RR, JO'R and VS. RHM and RCC produced the manuscript and all authors contributed to the final version.

**Competing interests**

The authors declare no competing interests.

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**Figure captions:**

**Figure 1 |Effects of differing prescribed fire frequencies on peat and C accumulation rates with respect to: (a) burn treatment and (b) number of burns applied.** Key for a. R = unburned since ca. 1923, N= burned in 1954, L = burned in 1954 and then every 20 years, S = burned in 1954 and then every 10 years; treatments denoted with similar small letters were not detected as significantly different (Tukey HSD, Peat = P<0.020; C = P<0.027); b. Linear regressions (±95% confidence limits are illustrated); equations (±SE) are presented in Supplementary Table S1.

**Figure 2 | GLM modelled responses of differing prescribed fire frequencies on community diversity and abundance of major species.** Abundance units are number of hits by pin quadrat[38,39]**.** a-c represent the effects of prescribed burning through time; d represents treatment effects as temporal effects were not significant. Key: N= 1-burn in 1954 (green, the intercept), L = 3-burns, burned in 1954 and every 20 years (blue), S = 6-burns, burned in 1954 and every 10 years (red). Significance: ns = not significant, P>0.05; + = P<0.05, +++/---, P <0.000; direction of effects are shown by + and – symbols.

**Figure 3| Summarised impacts of the four fire return intervals on key ecosystem properties|** a. Species composition, the arrows reflect relative increases and the figures are the final mean frequencies of key species, b. Carbon in the above-ground biomass, c. Peat and C net accumulation rates, and d. mass of C the surface 1 cm and 5 cm peat.

## METHODS

**Description of the Moor House Experiment and sampling protocol**. Moor House National Nature Reserve (NNR) is located in the Northern Pennines of England, and covers 40 $km^2$ of upland blanket bog, the largest area of ombrotrophic, mire-covered moorland in England[50]. The management pressure on this reserve is very low; there has been no burning outside this experiment for ca. 100 years and is approaching the lower end of the natural burn return cycle for unmanaged peatlands in upland England (ca. 115-250 years[12-13]). Sheep-grazing pressure on blanket bog is low; it was ca. 0.5 sheep $ha^{-1}$ when 15,400 sheep grazed the entire reserve pre-1970, and since then there has been a reduction to ca. 7,000 in 1970 and 3,500 after 2001. Moreover, the sheep grazing pressure is mainly concentrated on grassland areas outside the blanket bog[51].

The Sheep-grazing and Burning Experiment was established at Hard Hill (British grid reference; NY 758 328; Latitude 54.689656, Longitude -2.376928) in 1954 to investigate the effects of low-density sheep grazing and long-term, prescribed burning on blanket bog vegetation. The experiment was set up with a randomized block, split-plot design with four blocks, each with two sheep-grazing treatments (background sheep grazing pressure versus no sheep grazing) applied randomly within block and the three prescribed burning sub-treatments applied randomly within sheep-grazing treatments (Supplementary Fig. S1). Both the sheep grazing and burning treatments are fixed effects within the experimental design. All the plots were burnt in 1954/5 (here denoted 1954), and thereafter, three prescribed burning treatments were applied: short-rotation, every 10 years (S); long-rotation, every 20 years (L); and no subsequent burn since 1954 (N). Each of the four blocks has an associated reference plot (R) which has not been burnt since at least 1923[38]; the plots are referred to by the number of burns implemented since 1954; R=0-burn, N=1-burn, L=3-burns, and S=6-burns. The burning treatments applied were intended to test the impacts of the prescribed burning in many areas of upland Britain that is routinely applied for moorland management. Historically, this management practice was implemented to increase sheep utilization of the available grazing, but more recently it has been used mainly to increase red grouse (*Lagopus lagopus scotica* Latham) numbers for sporting purposes[38,39,42]. The intention is to use fire to open up the canopy of the dominant shrub species (*Calluna vulgaris* (L.) Hull), then allowing it to regenerate from both seedlings and burned stems through a distinct post-fire succession[42,43,52a]. This management is carried out on rotation across the landscape, providing a mosaic of burned patches[17]. In the uplands, prescribed burning must by law be done between October 1$^{st}$ and 15$^{th}$ April[53]. At Moor House, burning is applied in late March or early April. However, as this site has very inclement weather[54] it often is not possible to burn on an exact schedule; thus burning is applied at the end of March or beginning of April in close as possible to the intended year[29,38-39]. The fires would be described as flaming fires[23,55] produced by "cool-burning"[56], and there is no evidence that smouldering peat fires have occurred[23]. Here, cores were only sampled from the grazed treatments as this is the "business-as-usual" management regime for most upland blanket bog in the UK[38-39].

**Field methods.** Following a pilot study in 2011 (not shown), two "Master" cores were sampled (July 2013) from the Reference plot of Block A (no burn since ca. 1923) for analysis of peat and C dry mass accumulation, air-fall Pb by XRF (Supplementary Fig. S2) and for radiometric dating (MH13/1, MH13/4, Supplementary Fig. S3). Comprehensive analysis of the peat and C dry mass accumulation rates was undertaken by sampling (June 2016) within each burning treatment with four cores from treatment R, eight cores from L and N and twelve cores from S; thus comprising 8 cores per block (1xR, 2xL, 2xN, 3xS) and 32 cores in total (MH16/1-32). Throughout, a hemi-cylindrical peat sampler (0.5 m x 0.05 m diameter) was used to extract the peat cores, and they were stored in guttering, sealed in plastic sleeves, and stored under refrigeration until analysis.

**Estimating down-core concentrations of air-fall PB.** Major element and trace metal concentrations (ppm) including air-fall Pb were determined on a wet sediment basis at 5mm resolution for each core using an Olympus Delta Energy Dispersive (ED)-XRF) mounted on a Geotek MSCL-XZ core scanner. The XRF has a 4 W Rhodium X-ray tube (8–40 keV; 5–200 µA excitement), a thermo-electrically cooled large-area silicon drift detector with the 6 mm diameter detector window covered with a thin (6 µm) polypropylene film to avoid contamination of the internal measurement sensors. Measurements were conducted in 'Soil' mode, which applies three successive X-ray intensities (15, 40 and 40 (filtered) keV beam conditions). The analyser undergoes daily standardisation procedures and is tested routinely using certified reference materials[57]. The measured uncertainties for Pb (µg $g^{-1}$) are around 1% at 100 ppm increasing to 25% at 5ppm, and so the variation through the peak airfall Pb from 1850-1940 are captured by the µXRF scanning. Repeat measurements of calibration materials, 16 dried hand-pressed powders, for Pb across concentrations ranging from 5 to 700 µg $g^{-1}$ produced average 2 sigma uncertainties of ±3 µg $g^{-1}$. For the objectives of this paper, the stable Pb measured by ED-µXRF the airfall pollutant concentrations are greater than 10 µg $g^{-1}$ throughout the period 1840 to 1960, therefore, our quantification is robust. For the deeper peats, Pb concentrations are closer to background and we struggled to detect plausible Pb data, with the exception of the spike association with Roman-age smelting dust from central Europe (0-400 AD).

**Radiometric dating the Master cores.** Here, we calibrated Pb deposition and hence peat growth using radioisotopic markers. The Master cores were sub-sampled at 1 cm intervals and bulk densities calculated using standard water displacement techniques and measurement of the wet and dry masses after freeze drying. Sub-samples from each core were analysed for $^{210}$Pb, $^{226}$Ra, $^{137}$Cs and $^{241}$Am by direct gamma assay in the Liverpool University Environmental Radioactivity Laboratory using a Canberra SAGe well-type coaxial low background intrinsic germanium detectors[58]. $^{210}$Pb was determined via its gamma emissions at 46.5 keV, and $^{226}$Ra by the 295 keV and 352 keV γ-rays emitted by its daughter radionuclide $^{214}$Pb following 3 weeks storage in sealed containers to allow radioactive equilibration. $^{137}$Cs and $^{241}$Am concentrations were estimated by their emissions at 662 keV and 59.5 keV respectively. The absolute efficiencies of the detectors were determined using calibrated sources and sediment samples of known activity. Corrections were made for the effect of self-absorption of low energy γ-rays within the sample[59]. The results were plotted alongside data for atmospheric fallout Pb and Zn concentrations measured by ED-XRF (Supplementary Fig. S3), with supported $^{210}$Pb activity assumed to be equal to the measured $^{226}$Ra activity, and unsupported $^{210}$Pb activity calculated by subtracting supported $^{210}$Pb from the measured total $^{210}$Pb activity.

**Core MH13/1.** Extrapolation of the total $^{210}$Pb data (Supplementary Fig. S3c) indicates that 99% equilibrium with the supporting $^{226}$Ra (corresponding to around 150 years accumulation) occurred at a depth of between 14-15 cm. Because of the very low $^{226}$Ra concentrations (mean value 4 Bq kg$^{-1}$) it was not practicable to continue total $^{210}$Pb measurements to a point where radioactive equilibrium was achieved fully. Although there were some irregularities in the unsupported $^{210}$Pb record (Supplementary Fig. S3b) concentrations declined more or less exponentially with depth, suggesting relatively uniform peat accumulation over the past 100 years or so. High $^{137}$Cs concentrations (Supplementary Fig. S3b) in the form of a double peak were detected in samples between 1 and 4 cm. The proximity to the surface of the core suggests that this feature records fallout from the 1986 Chernobyl accident. Downward migration of Chernobyl $^{137}$Cs appears to have masked any evidence of an earlier $^{137}$Cs peak recording the 1960s fallout maximum from the atmospheric testing of nuclear weapons. Traces of $^{241}$Am (Supplementary Fig. S3b), also a product of nuclear weapon test fallout[60] in the late 1950s and early 1960s, were however, detected in samples between 3-8 cm. The $^{210}$Pb chronology calculated using the CRS model[56] places 1986 at around 3 cm and 1963 at around 6 cm, which shows a reasonable degree of consistency between these two independent dating methods. Calculations using the alternative CIC $^{210}$Pb model gave results broadly similar to those determined from the CRS model, confirming the suggestion that net peat accumulation rates have not change significantly over the past century. Given the large uncertainties in both the $^{210}$Pb and $^{137}$Cs records the mean accumulation rate, $0.010 \pm 0.002$ g cm$^{-2}$ yr$^{-1}$ (0.10 cm yr$^{-1}$), was used to calculate the age-depth model (Supplementary Fig. S3).

**Core MH13/4.** The total $^{210}$Pb record in this core was broadly similar to that in MH1, though a significantly greater 99% equilibrium depth (estimated to be around 22 cm) suggests a significantly greater peat accumulation rate at the site of this core. Although unsupported $^{210}$Pb concentrations (Supplementary Fig. S3c) vary irregularly with depth, since the overall decline is again more or less exponential, it appears that there have been no major changes in the net peat accumulation rate (Supplementary Fig. S3d). High $^{137}$Cs concentrations (Supplementary Fig. S3b) above 4 cm probably originate from 1986 Chernobyl fallout, whilst traces of $^{241}$Am present in samples above 9 cm most probably originate from fallout from the atmospheric testing of nuclear weapons. However, in neither case are there distinct features that can be linked clearly to specific dates. The $^{210}$Pb chronology was calculated using the CRS model[61], and although a lack of clarity in the $^{137}$Cs/$^{241}$Am records prevented close validation of the $^{210}$Pb calculations, since these place 1986 at around 5 cm and 1963 at around 9 cm the two methods are broadly consistent. Use of the CIC model yielded similar results to those given by the CRS model, supporting the suggestion that net peat accumulation rates have been relatively constant. The age-depth model (Supplementary Fig. S3d) was calculated using the mean value of $0.017 \pm 0.003$ g cm$^{-2}$ yr$^{-1}$ (0.17 cm yr$^{-1}$).

**Calculating peat and C accumulation rates (Cores M16/1-32).** Peat accumulation rates were derived using features or markers in the pronounced down-core atmospheric fall-out stable Pb profile measured by XRF. Pb is relatively immobile in ombrotrophic peat and has produced profile repeatable between all the cores[62]. Four good age markers were detected and assigned ages from the radiometric dating at 1876, 1963, 1986 and the peat surface (2016). As 1963 was the closest to the start of the Hard Hill experiment this marker was used to estimate recent peat and C accumulation rates. Peat growth rates (cm yr$^{-1}$) were calculated for each core across the two periods (1876-1963 and 1963-2016), essentially pre- and post-experiment. C accumulation was measured for the peat sequence using Near-Infra-Red Spectrophotometry (NIRS) cross-calibrated using a training set of direct mass loss-on-ignition (l-o-i) measurements. NIRS results have been shown to correlate strongly with the organic content of sediments[63-65]. NIRS reflectance was measured on each 1-cm depth samples from all cores using a BRUKER MPA FT-NIR spectrometer; lightly-ground peat was scanned at 4 nm intervals between 3598-12493 nm. L-o-i was measured on each 1-cm depth section from four cores, one selected form each burning treatment; peat samples were ashed at 550˚C for 3 h[63]. Cross-calibration indicated a strong correlation ($r^2$= 86%) between the first derivative of the entire NIR spectra and measured l-o-i (Supplementary Fig. S4). L-o-i and hence C concentration (as a normative 40% of the burnt mass loss) was predicted from the NIRS data. This NIRS-based approach provides robust, rapid and non-destructive estimates for l-o-I and C concentrations. The C accumulation rate (g C $m^2$ $yr^{-1)}$ was calculated using the measured or NIRS predicted l-o-I results for each core for the periods 1876-1963 and 1963-2016.

**Statistical Methods.** All analyses were performed in the R statistical environment[66]; three hypotheses were tested with respect to peat accumulation. (1) The peat and C mass accumulation rates were similar in the pre-burn (1876-1963) and post-burn (1963-2016) periods; here pre- and post-burn rates from the 0-burn treatments were compared using a Student's t-test (function 't.test', untransformed data). (2) Prescribed burning implemented within the experiment changed peat and C mass accumulation rates. Here, effects of the prescribed burning treatments on accumulation rates since 1963 were tested using analysis of variance (functions 'aov' and 'TukeyHSD', $log_e$ transformation). (3) Peat and C mass accumulation rates are dependent on different prescribed burning frequencies. Here, the relationships between accumulation rates of peat depth and C since 1963 were assessed using simple linear regression ('lm' function, untransformed data). For hypotheses 2 and 3, QQ-plots were inspected to ensure normality; in the linear regression analysis transformations did not improve the analysis, so analyses based on raw data are presented.

To estimate the time taken to recover the C lost after wildfire, we calculated the total amount of C in both the surface vegetation and surface peat at two depths (0-1 cm and 0-5 cm) and divided by the C accumulation rate measured for the 6-burn treatment. We used a randomization approach (n=10,000) selecting data from each of the three variables (mean and SD) using the ‘rnorm’ function and calculating the mean and 95% confidence limits (‘quantile’ function). The mean values (±SD) were: vegetation C = 820±127 g C $m^{-2}$; $Peat_{0\text{-}1cm}$ C = 240±22 g C $m^{-2}$; $Peat_{0\text{-}5cm}$ C= 1274±82 g C $m^{-2}$and C accumulation rate =36±2.6 g C $m^{-2}$ $yr^{-2}$ (6-burn value).

In addition, in order to provide ancillary information about the effects of prescribed burning on the moorland community, data on species frequency of occurrence, derived from pin-quadrats) were abstracted from the vegetation monitoring program for this experiment (1972-2013)[29]. Here, modelled responses, derived from a GLM analysis for Shannon-Weiner diversity index and the frequency of occurrence of the major components of the vegetation (*C. vulgaris*, *Eriophorum vaginatum* (L.); both Poisson error distribution, and combined *Sphagnum* (L.) spp. Binomial error distribution). Only the modelled responses of the ungrazed treatments are presented for the N, L and S treatments; comparable data for R were not collected.

**Data availability**. The data that support the findings of this study are available in (1) DataCat: the University of Liverpool Research Data Catalogue with the identifier [http://dx.doi.org/10.17638/datacat.liverpool.ac.uk/531] for peat and C accumulation rates[66], and (2) the NERC Environmental Information Data Centre with the identifier https://doi.org/10.5285/0b931b16-796e-4ce4-8c64-d112f09293f7 for species change[67].

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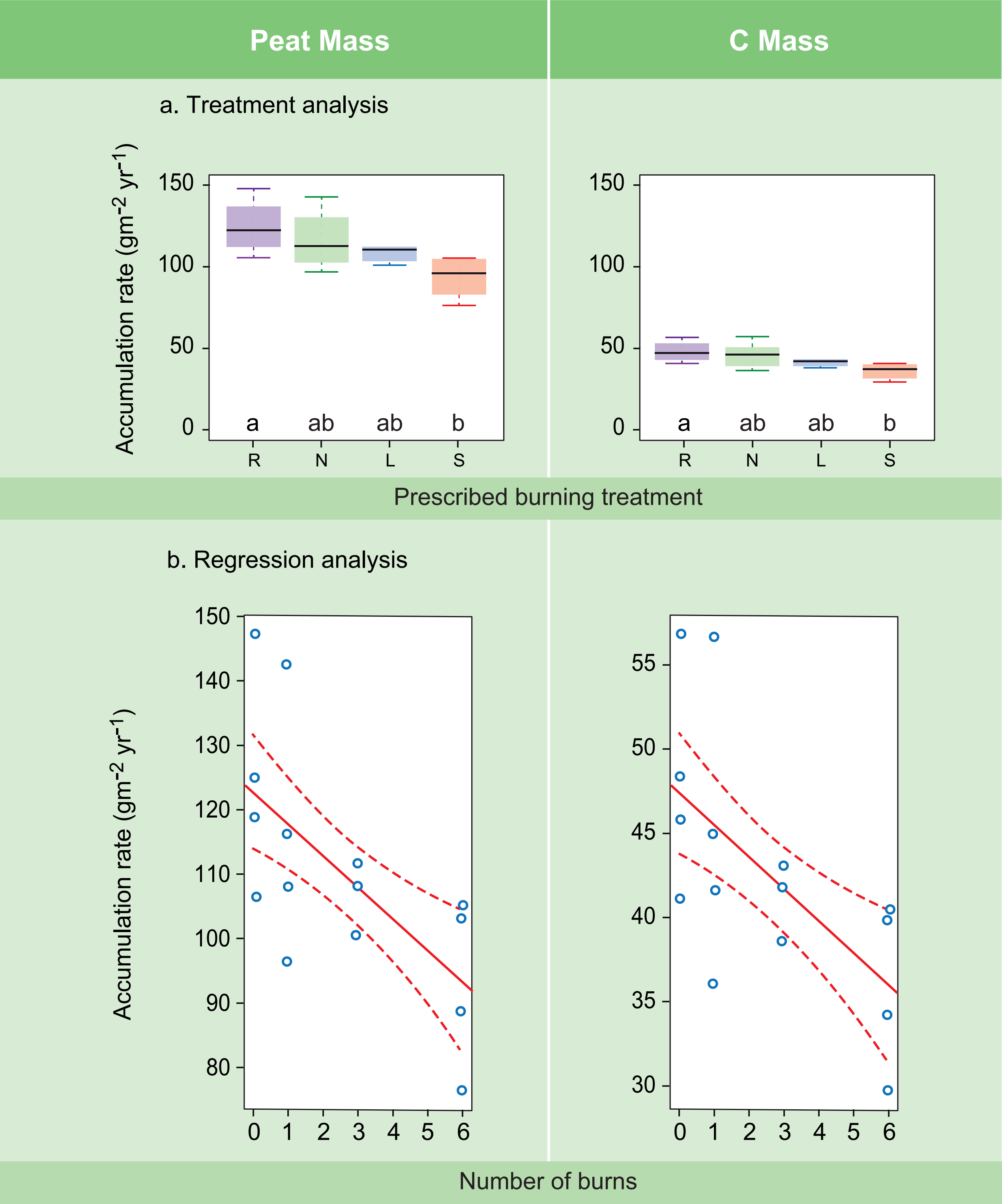
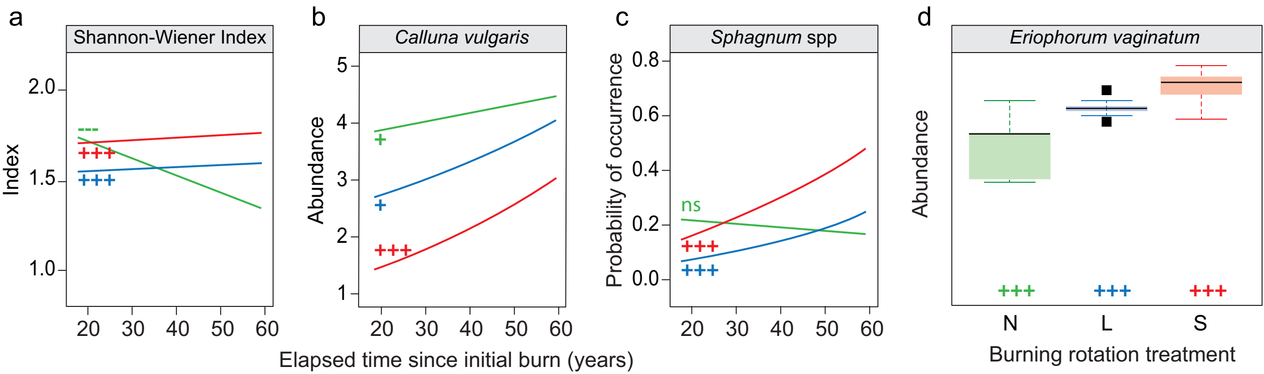
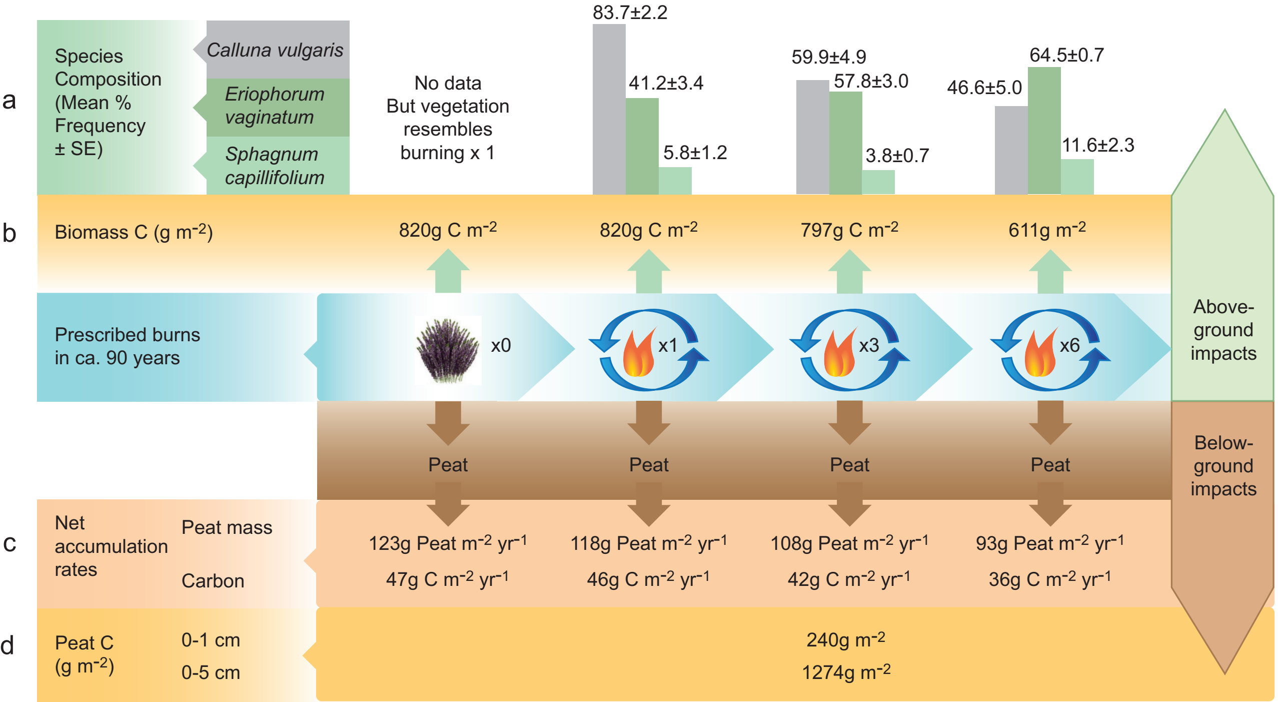


Peat Mass
C Mass
a. Treatment analysis
Accumulation rate (gm-2 yr-1)
150
100
50
0
a
ab
ab
b
R
N
L
S
Prescribed burning treatment
b. Regression analysis
Accumulation rate (gm-2 yr-1)
150
140
130
120
110
100
90
80
55
50
45
40
35
30
0
1
2
3
4
5
6
Number of burns



a
Shannon-Wiener Index
Index
2.0
1.5
1.0
---
+++
+++
b
Calluna vulgaris
Abundance
5
4
3
2
1
+
+
+++
c
Sphagnum spp
Probability of occurrence
0.8
0.6
0.4
0.2
0.0
ns
+++
+++
20 30 40 50 60
Elapsed time since initial burn (years)
d
Eriophorum vaginatum
Abundance
+++ +++ +++
N L S
Burning rotation treatment



a
Species Composition (Mean % Frequency ± SE)
Calluna vulgaris
Eriophorum vaginatum
Sphagnum capillifolium
No data
But vegetation resembles burning x 1
83.7±2.2
41.2±3.4
5.8±1.2
59.9±4.9
57.8±3.0
3.8±0.7
46.6±5.0
64.5±0.7
11.6±2.3
b
Biomass C (g m-2)
820g C m-2
820g C m-2
797g C m-2
611g m-2
Prescribed burns in ca. 90 years
x0
x1
x3
x6
Above-ground impacts
Peat
Peat
Peat
Peat
Below-ground impacts
c
Net accumulation rates
Peat mass
Carbon
123g Peat m-2 yr-1
47g C m-2 yr-1
118g Peat m-2 yr-1
46g C m-2 yr-1
108g Peat m-2 yr-1
42g C m-2 yr-1
93g Peat m-2 yr-1
36g C m-2 yr-1
d
Peat C (g m-2)
0-1 cm
0-5 cm
240g m-2
1274g m-2