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Prescribed burning, atmospheric pollution and grazing effects on peatland vegetation composition

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Abstract

1. Peatlands are valued for ecosystem services including carbon storage, water provision and biodiversity. However, there are concerns about the impacts of land management and pollution on peatland vegetation and function.
2. We investigated how prescribed vegetation burning, atmospheric pollution and grazing are related to vegetation communities and cover of four key taxa (*Sphagnum* spp., *Calluna vulgaris*, *Eriophorum vaginatum* and *Campylopus introflexus*) using two datasets from a total of 2,013 plots across 95 peatland sites in the UK.
3. Non-metric multidimensional scaling and permutational multivariate analysis of variance showed differences in vegetation community composition between burned and unburned plots at regional and national scales.
4. Analysis showed that burned sites had less *Sphagnum* and greater *C. vulgaris* cover on a national scale. On a regional scale, plots burned between 2 and 10 years ago had greater cover of invasive moss *C. introflexus* and less *E. vaginatum* than unburned sites.
5. Livestock presence was associated with less *Sphagnum* and *C. vulgaris*, while atmospheric pollution was associated with less *Sphagnum*, but greater *C. introflexus* cover, and appeared to have more impact on burned sites.
6. *Synthesis and applications*. Burning, grazing and atmospheric pollution are associated with peatland vegetation composition and cover of key species, including *Sphagnum*. We suggest that, to promote cover of peat-forming species, peatlands should not be routinely burned or heavily grazed. Current or historical atmospheric pollution may hinder peat-forming species, particularly on burned sites.

KEYWORDS
fire, grazing, land management, nitrogen, peatlands, peat moss, pollution, prescribed burning, sulphur, vegetation

1 | INTRODUCTION

Peatlands cover an estimated 4 million km^2 globally (Yu, Loisel, Brosseau, Beilman, & Hunt, 2010), and are important for carbon storage, hydrological function and biodiversity (Evans et al., 2014). Plant community composition impacts these functions (e.g. Kivimaki, Yli-Petays, & Tuittila, 2008), so knowledge of how peatland vegetation responds to environmental change is vital. Arguably the most important plant for peatland function globally is *Sphagnum*, a genus of mosses which retain water and acidify their local environment, slowing decomposition and enabling peat accumulation (Clymo & Hayward, 1982). On sloping blanket peatlands, *Sphagnum*-dominated surfaces experience slower overland flow than sedge-dominated or bare peat surfaces (Holden et al., 2008), which may lead to reduced peak flows in rivers (Gao, Holden, & Kirkby, 2015).

Due to the carbon storage and hydrological functions of *Sphagnum*, its re-establishment is often a focus in peatland restoration (Evans et al., 2014; Rochefort, 2000). Sedges such as *Eriophorum vaginatum* also form peat, and can colonise and stabilise eroding peat (Evans & Warburton, 2008). Dwarf shrubs such as *Calluna vulgaris* occur naturally on blanket peatlands, often as a short canopy (Averis et al., 2004), but dominance can impact peatland soil ecosystem functions such as net carbon storage (Dixon, Worrall, Rowson, & Evans, 2015), water storage and routing (Holden, 2005) and dissolved organic carbon production (Armstrong et al., 2010). Invasive species including *Campylopus introflexus*, a moss native to the southern hemisphere which is now widely distributed in several European countries (Equihua & Usher, 1993), may also threaten ecosystem function by competing with native species.

Fire is common in peat-rich biomes and can threaten carbon stocks (Turetsky et al., 2015). Prescribed burning occurs on many peatlands globally for purposes including wildfire prevention, forestry, land clearance and habitat management (Brown, Palmer, Johnston, & Holden, 2015; Buytaert et al., 2006). Unlike wildfires, prescribed burns are usually controlled to ignite vegetation, but not the underlying peat. In the UK, patches of up to 4000 m^2 are burned in rotations of 5–20 years to create a mosaic of *C. vulgaris* ages to improve forage and nesting habitat for the game bird red grouse (*Lagopus lagopus scotica*). Despite guidance against burning on deep peat in the UK (Defra, 2007; Scottish Government, 2011; Welsh Assembly Government, 2008), there is evidence that burns have increasingly encroached onto these areas (Douglas et al., 2015; Yallop et al., 2006). In recent years, there has been considerable debate over burning (Brown, Holden, & Palmer, 2016; Dougill et al., 2006), and further evidence of how it affects vegetation is required to inform management and policy.

Fire can affect vegetation by both combustion and alteration of local environmental conditions. For example, greater peat bulk density (Holden et al., 2015) and lower near-surface hydraulic conductivity (Holden et al., 2014) in recently burned areas may reduce water availability to *Sphagnum*, which relies on passive capillary transport (Thompson & Waddington, 2008). Burning can cause higher maximum and lower minimum near-surface soil temperatures in the years following a fire (Brown et al., 2015), which may impact *Sphagnum* growth negatively (Walker, Ward, Ostle, & Bardgett, 2015). Interactions between vegetation properties, fire severity and local soil conditions may explain why past evaluations of burning impacts on *Sphagnum* have yielded mixed conclusions (Glaves et al., 2013; Worrall, Clay, Marrs, & Reed, 2010). *Calluna vulgaris* shows evidence of fire adaptation and may increase after burning unless suppressed by grazing. Burning may also allow new species to colonise. *C. introflexus* can carpet bare soil rapidly (Southon, Green, Jones, Barker, & Power, 2012) but whether this occurs to an ecologically significant extent after burning is unknown. One issue that prevents generalisations from being made is that the current evidence base on the effects of burning on peatland vegetation draws mainly from localised case studies, and the applicability of these findings to larger scale (e.g. national) patterns remains unestablished.

When considering peatland vegetation patterns at large scales (i.e. national to global), atmospheric pollution becomes an important consideration. *Sphagnum* dieback near industrial centres has been attributed to sulphur deposition (Ferguson, Lee, & Bell, 1978). Nitrogen additions can limit *Sphagnum* growth directly (Granath, Strengbom, & Rydin, 2012) or by increasing competition (Malmer, Albinsson, Svensson, & Wallén, 2003), and can favour *C. introflexus* (Field et al., 2014; Southon et al., 2012). Grazing is also widespread on many peatlands and experimental work has shown varying responses among plant species (Milligan, Rose, & Marrs, 2016). Further knowledge of the relationship between grazing and peatland vegetation on a national scale may help when weighing up economic, cultural and conservation concerns.

The UK has approximately 1.5 million ha of blanket peatland (Bain et al., 2011); around 13% of the global total. The extent to which this peatland is affected by widespread management practices (burning and grazing), atmospheric pollution, and interactions between these drivers is not clear. In the UK, prescribed vegetation burning and atmospheric pollution both intensified during the 20th century, so knowledge of their interactive effects is valuable.

In this paper, we present the first synthesis of national scale peatland vegetation survey data from 1893 plots across 85 sites in England, alongside a regional dataset with 123 plots across 10 sites. The study aims to evaluate relationships between environmental drivers (including prescribed burning, atmospheric pollution and grazing) and blanket peatland vegetation community composition, as well as abundance of four key taxa (*Sphagnum* spp., *C. vulgaris*, *E. vaginatum* and *C. introflexus*). The findings are discussed in the context of peatland restoration and management.

2 | MATERIALS AND METHODS

2.1 | Data sources

Two datasets were analysed (Figure 1). The first comprised vegetation survey data from the UK NERC-funded EMBER (Effects of Moorland Burning on the Ecohydrology of River basins) project, which established 12–15 sampling plots evenly distributed between top, mid and toe slope positions in each of five unburned and five burned catchments in the English Pennines (Table 1) (Brown, Holden, & Palmer, 2014). Within each burned site, plots were distributed between four age classes; burned <2 years (B2), 3–4 years (B4), 5–7 years (B7) and >10 years (B10+) prior to the study. The Domin abundance of vascular plants, bryophytes and lichens within a 2 m × 2 m quadrat in each plot was recorded and transformed to an approximation of percentage cover using the Domin 2.6 transformation (Currall, 1987). Peat depth was measured up to a 118 cm probe length limit. National Vegetation Classification (NVC) types (Rodwell, 1991) present at each site were also determined.

The second source was a habitat condition monitoring dataset (henceforth CM data). This was derived from a representative

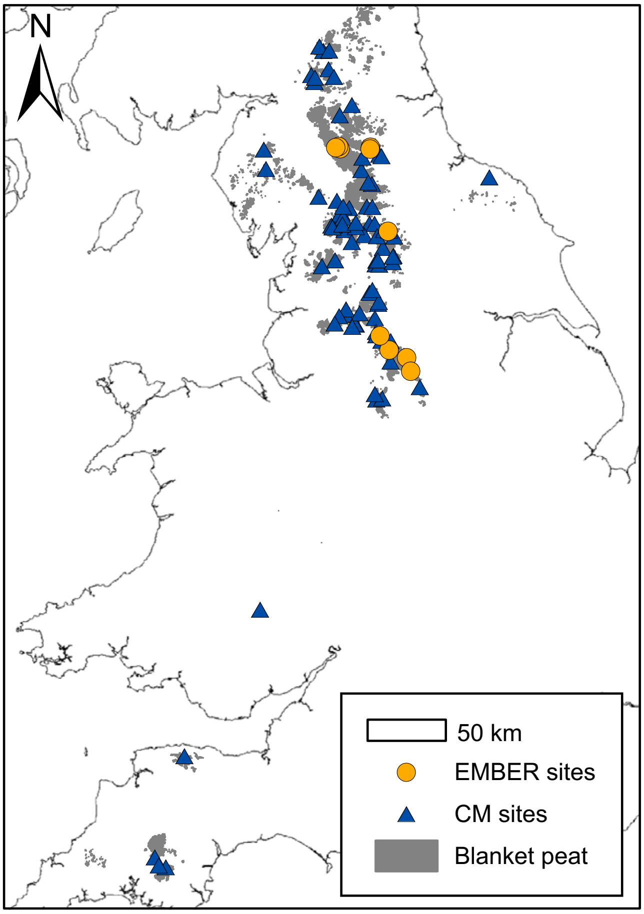

FIGURE 1 Map of study sites and the extent of blanket peatland mapped as part of Natural England's Priority Habitat Inventory © Natural England copyright with Ordnance Survey data © Crown copyright 2017 [Colour figure can be viewed at wileyonlinelibrary.com]

sample of mapped habitat polygons in Natural England's Blanket Bog Priority Habitat Inventory. Percentage cover values for indicator species (Table S1) were recorded using 2 m × 2 m quadrats. Peat depth was measured up to a 150 cm probe length limit and the presence of livestock droppings was recorded. Data from 1,893 plots (distributed randomly within 85 sites) with peat depth ≥30 cm were used in this analysis; 30 cm representing the minimum depth on which blanket peatland vegetation normally occurs (Lindsay, 2010). Satellite imagery from Google Earth™ and Bing Maps™ was used to determine whether plots had been recently burned (788, 41.6%) or not (1105) based on the visibility of burn patches. This is an effective method for determining the burn status of areas of peatland (Allen, Denelle, Ruiz, Santana, & Marrs, 2016; Yallop et al., 2006). However, in this instance, it was not possible to determine the age of individual burn patches, so the 'burned' category encompassed a range of burn ages. The time taken for visual recovery after burning varies according to vegetation regeneration, but is usually at least 25 years where dwarf shrubs are abundant (Yallop et al., 2006). Detection of burning on grass dominated moorland from imagery is problematic due to a short-lived burn signature (Yallop et al., 2006), so 303 plots with grassland vegetation types (14% of 2,196 total plots) were excluded from the analysis.

Atmospheric pollution data (deposition of acid, nitrogen, ammonia, nitrogen oxides, sulphur dioxide and ozone) were sourced from the Air Pollution Information Service database (APIS, 2014), which provides 5-km resolution modelled values averaged over 3–5 years (ozone 2007–2012, all other pollutants 2012–2014). Elevation and slope values were extracted from the Ordnance Survey Terrain 50 digital elevation model using ESRI ArcGIS 10.2.2.

TABLE 1 Locations and dominant national vegetation classification types of EMBER (Effects of Moorland Burning on the Ecohydrology of River basins) sites; all occur on blanket peatland

Site	Region	NVC type(s)
Burned		
Bull Clough	Peak District	H9b (*Calluna vulgaris–Deschampsia flexuosa* heath), M6 (*Carex echinata–Sphagnum recurvum/auriculatum* mire)
Eggleshope Beck	North Pennines	M19a (*Calluna vulgaris–Eriophorum vaginatum* blanket mire)
Lodgegill Sike	North Pennines	M6, H12a (*Calluna vulgaris–Vaccinium myrtillus* heath), M19a
Rising Clough	Peak District	H9b
Woo Gill	Yorkshire Dales	M19a, M20b (*Eriophorum vaginatum* blanket and raised mire)
Unburned		
Green Burn	North Pennines	M19b (*Calluna vulgaris–Eriophorum vaginatum* blanket mire)
Moss Burn	North Pennines	M19b
Trout Beck	North Pennines	M19b
Crowden Little Brook	South Pennines	M20b
Oakner Clough	South Pennines	M20b

2.2 | Data analysis

Statistical analyses were carried out in R 3.1.1 (R Development Core Team, 2010) with the packages ggplot2 (Wickham, 2009), lme4 (Bates, Maechler, Bolker, & Walker, 2015), MASS (Venables & Ripley, 2002) and vegan (Oksanen et al., 2013). To explore how vegetation community composition differs according to burn status, non-metric multidimensional scaling (NMDS), an ordination method which projects multivariate data into a space with fewer dimensions, was performed on each dataset using Bray–Curtis dissimilarities and the best 2D fits were retained. This enabled comparison of burned and unburned plots, with more similar vegetation communities appearing closer together. Permutational multivariate analysis of variance (PERMANOVA) was used to test for significant differences between vegetation communities in plots with different burn ages. The envfit function in R was used to fit linear relationships between environmental variables and NMDS axes to enable exploration of their associations with vegetation community composition.

Relationships between cover of key taxa (*Sphagnum* spp., *C. vulgaris*, *E. vaginatum* and *C. introflexus*) and environmental variables were investigated using generalised linear mixed models (GLMMs) for each dataset separately, with Poisson error distributions and log links as the percentage cover data were bounded at zero. Site was included in all models as a random effect to account for grouping of plots within sites in both datasets. Species cover data from both datasets were overdispersed, which was accounted for by including an observation level random effect in the models (Harrison, 2014). Continuous variables were tested for correlation with each other and where a significant ($p < .05$) relationship with a Pearson correlation coefficient >0.2 occurred between two variables, one was excluded (Tables S2 and S3) according to which was considered most ecologically relevant. The remaining continuous variables were scaled and centred.

Two sets of GLMMs were fitted separately to each dataset to enable model convergence and facilitate interpretation of the results. First, models with all non-correlated predictor variables identified as significant in the envfit analysis, along with burn status (and livestock presence for CM data) were fitted. Second, models with burn status, nitrogen deposition and their interaction were fitted. The interaction models investigated the effect of nitrogen deposition on vegetation cover within and between burn groups. R^2 values were calculated using the methods of Nakagawa and Schielzeth (2013).

3 | RESULTS

3.1 | Vegetation community composition

Non-metric multidimensional scaling of the EMBER data showed separation of burned and unburned plots (Figure 2), but no clear pattern within burned plots related to time since burning. Permutational multivariate analysis of variance indicated significant effects of site (df = 9,106, R^2 = 0.40, $p < .001$) and burn status (all levels included, df = 4,106, R^2=0.14, $p < .001$) on species composition. When only burned plots were compared, the effect of burn age was not significant (p = .06, df = 3,112). The distribution of sites in the NMDS plot reflects both location and NVC type (Table 1). Within the unburned sites, two groups are apparent (Figure 2a, Figure S1); North Pennine sites with NVC type M19b *C. vulgaris* and *E. vaginatum* mire (Green Burn, Moss Burn and Trout Beck) and South Pennine sites with M20b *E. vaginatum* mire (Oakner and Crowden). Associations for the burned sites were less clearly distributed; Eggleshope Beck (North Pennine, M19a) is similar to the unburned North Pennine sites, whereas Lodgegill Sike (North Pennine M6, H12a, M19a) has more in common with Rising Clough and Bull Clough, both Peak District sites with H9b vegetation, a *Calluna*-dominated dry heath type characteristic of frequent burning (Elkington, Dayton, Jackson, & Strachan, 2001). Woo Gill (Yorkshire Dales, M19a, M20b) lacks obvious similarity to other sites. Peat depth, elevation and six atmospheric pollution variables were associated with species composition (Figure 2).

In the NMDS ordination of the CM data, unburned and burned plots occupy overlapping areas in the ordination space (Figure 3), but with a greater concentration of unburned and burned sites in negative and positive regions of axis 1 respectively. PERMANOVA indicated a significant difference between burned and unburned plots (df = 1,1892, R^2 = 0.05, $p < .001$). Geographic location (northing and easting), physical site attributes (slope, elevation and peat depth), browsing and five atmospheric pollution variables were all associated with species composition (Figure 3).

3.2 | Relationships between key taxa and environmental variables

Results from the GLMM models revealed that several environmental and management variables were correlated with cover of key taxa (Figure 4, Table S4). At EMBER sites, differences in *Sphagnum* and *C. vulgaris* cover between unburned plots and plots of different burn ages were not significant, but there was less *E. vaginatum* on B2 and B7 plots than unburned plots. Mean *E. vaginatum* cover values were 11.8% and 4.9% for B7 and B2 plots compared to 18.1% on unburned plots. Conversely, there was more *C. introflexus* on B2, B4 and B7 plots with mean cover values of 23.9%, 21.9% and 18.3%, respectively, compared to 0.5% on unburned plots. Nitrogen deposition in the EMBER dataset was negatively correlated with *Sphagnum* and positively correlated with *C. introflexus*. Increasing peat depth was associated with more *E. vaginatum*, while increasing elevation was associated with more *Sphagnum*, but less *C. introflexus*.

The CM data indicated less *Sphagnum* on burned plots with a mean cover of 8.2% compared to 10.5% on unburned plots, and also on plots with livestock droppings with a mean cover of 6.7% compared to 10.7% on those without (Figure 5, Table S4). There was more *C. vulgaris* on burned plots with 37.5% mean cover compared to 10.7% on those without, and less *C. vulgaris* on plots with livestock droppings with a mean of 12.3% cover compared to 24.4% on plots without droppings. Plots further north were associated with more *Sphagnum*, *C. vulgaris* and *E. vaginatum*, while elevation was positively correlated with *C. vulgaris* and *E. vaginatum* but negatively correlated with *Sphagnum*. Nitrogen deposition had a negative relationship with *Sphagnum*.

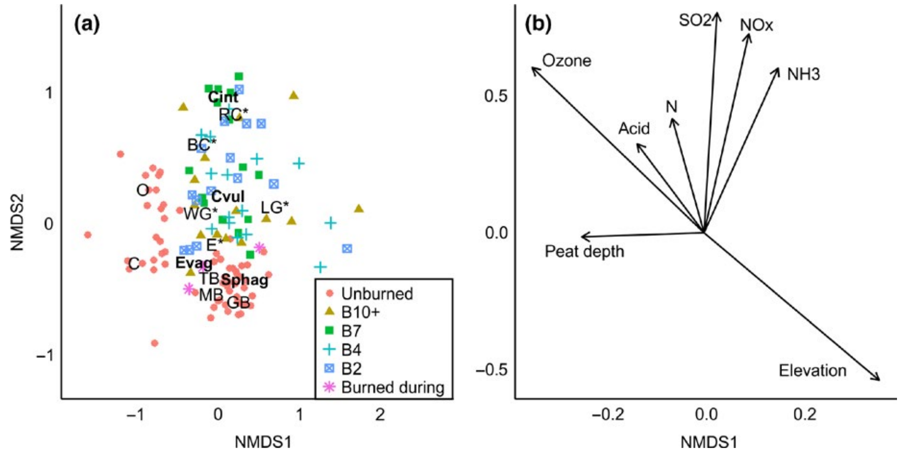

FIGURE 2 Non-metric multidimensional scaling (NMDS) ordination (stress = 0.23) of effects of moorland burning on the ecohydrology of river basins (EMBER) vegetation data showing (a) unburned plots (n = 60) and plots burned 2, 4, 7 or 10+ years prior to survey (all n = 15) with centroids for site (BC, Bull Clough; E, Eggleshope Beck; LG, Lodgegill Sike; RC, Rising Clough; WG, Woo Gill; GB, Green Burn; MB, Moss Burn; TB, Trout Beck; C, Crowden; O, Oakner; sites with * are burned) and species of interest (Cint, *Campylopus introflexus*; Cvul, *Calluna vulgaris*; Evag, *Eriophorum vaginatum*; Sphag, *Sphagnum* spp.), (b) Envfit vectors for environmental variables which were significantly correlated (p < .05) with the NMDS ordination (direction of arrow = increasing value of variable, length of arrow = strength of correlation) [Colour figure can be viewed at wileyonlinelibrary.com]

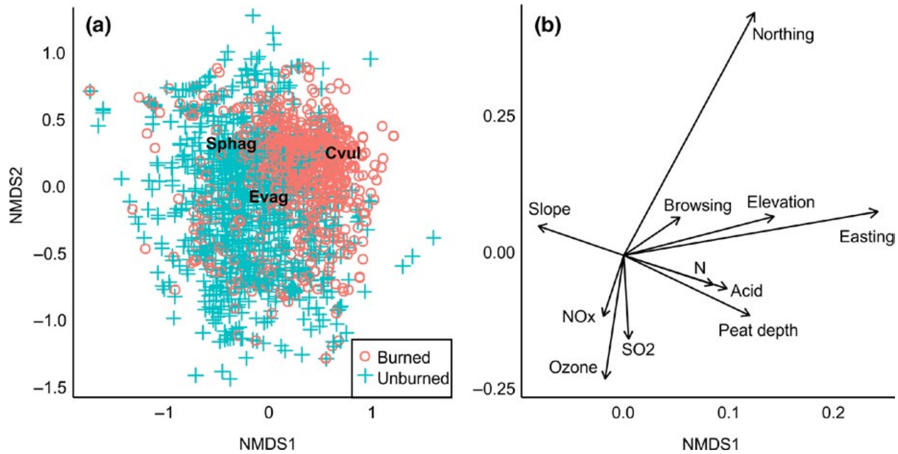

FIGURE 3 Non-metric multidimensional scaling (NMDS) ordination (stress = 0.24) of condition monitoring (CM) vegetation data showing (a) unburned (n = 788) and burned plots (n = 1,105) and centroids for species of interest (Cvul, *Calluna vulgaris*, Evag, *Eriophorum vaginatum*, Sphag, *Sphagnum* spp.) and (b) Envfit vectors for environmental variables which were significantly correlated (p < .05) with the NMDS ordination (direction of arrow = increasing value of variable, length of arrow = strength of correlation) [Colour figure can be viewed at wileyonlinelibrary.com]

3.3 | Interactions between burning and atmospheric pollution

At EMBER sites, an interaction between burn age and nitrogen deposition impacted cover of *E. vaginatum*, with a significantly more negative relationship on B2 plots than unburned plots (Figure 6, Table S5). In the CM data, nitrogen deposition had a significantly more negative relationship with *Sphagnum* cover on burned plots than unburned plots (Figure 7, Table S5).

4 | DISCUSSION

Analysis of the two datasets revealed that blanket peat vegetation community composition is associated with burning status, grazing, atmospheric pollution and site physical attributes including northing and elevation. The four taxa of interest in this study were associated with several of these variables. Model R^2 values indicate that our predictors explained more variation in the regional EMBER data than the national CM data (Figures 4–7), which is likely due to less variation in other environmental factors at the regional scale. The unexplained variation may be due to factors including management history and hydrological variables not captured by our predictors (e.g. drainage). Although effect size and significance differed between the two datasets, the direction of effects was generally consistent. The results suggest that burning, grazing and atmospheric pollution have the capacity to shift vegetation community composition on peatlands, and that burning and atmospheric pollution have an interactive effect on the abundance of key taxa in some cases.

4.1 | Drivers of community composition

Analyses spanning regional to national scales both suggested that burning is associated with differences in vegetation community

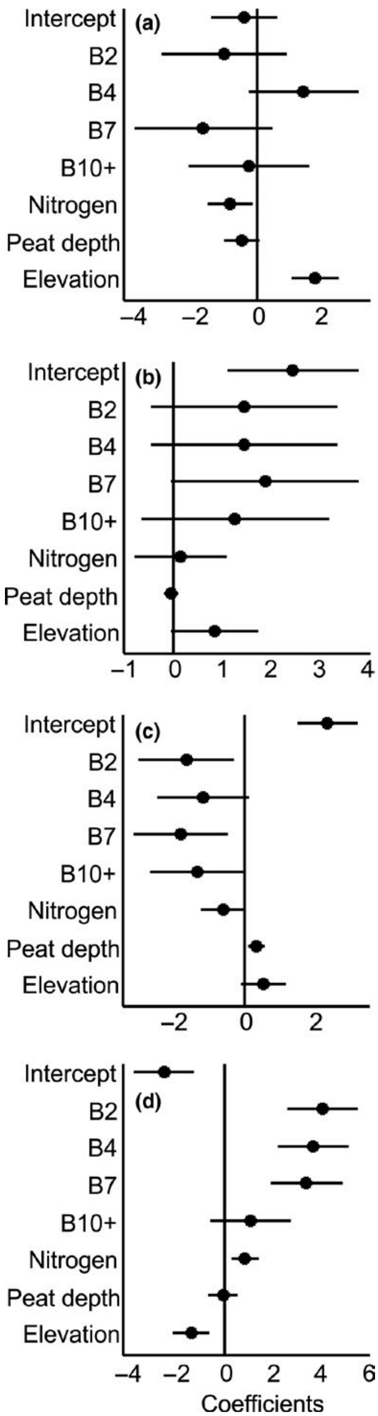

FIGURE 4 Coefficient plots of effects of moorland burning on the ecohydrology of river basins (EMBER) generalised linear mixed models (GLMMs) showing standardised coefficient estimates and 95% confidence intervals for comparison of (a) *Sphagnum*, (b) *Calluna vulgaris*, (c) *Eriophorum vaginatum* and (d) *Campylopus introflexus* cover on B2, B4, B7 and B10+ plots (all n = 15) to unburned plots (n = 60) and the relationships of cover with nitrogen deposition, peat depth and elevation. Marginal R^2 = 0.47, 0.30, 0.50 and 0.66 respectively; Conditional R^2 = 0.50, 0.90, 0.75 and 0.69 respectively

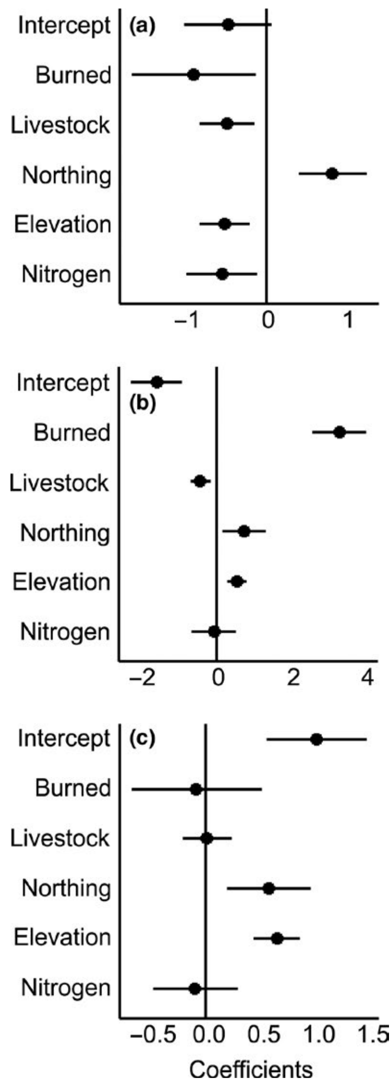

FIGURE 5 Coefficient plots of condition monitoring (CM) generalised linear mixed models (GLMMs) showing standardised coefficient estimates and 95% confidence intervals for the comparison of (a) *Sphagnum*, (b) *Calluna vulgaris* and (c) *Eriophorum vaginatum* cover on burned plots (n = 788) to unburned plots (n = 1,105); plots with livestock droppings (n = 544) to those without (n = 1,349); and the relationships of cover with northing, elevation and nitrogen deposition. Marginal R^2 = 0.13, 0.25 and 0.09 respectively; Conditional R^2 = 0.42, 0.73 and 0.50 respectively

composition, as previously observed in local (Hobbs, 1984; Lee et al., 2013) and regional (Harris et al., 2011) studies. The similar envfit relationships of different atmospheric pollutants may be due to comparable effects on vegetation or spatial correlation in deposition and thus it is difficult to evaluate pollutants individually. Blanket peatlands are ombrotrophic, nutrient-poor habitats so it is likely that input of nutrients via pollution impacts species composition by enhancing the competitive ability of vascular plants as well as via toxic effects on pollution sensitive species (Berendse et al., 2001).

The NMDS and PERMANOVA analysis of the EMBER data indicated that burned plots of different ages were more similar to each other than to unburned plots, suggesting that vegetation was still distinct from unburned sites 10+years after burning. Full recovery may depend on re-establishment of features including microtopography, thermal regime and hydrological function, which

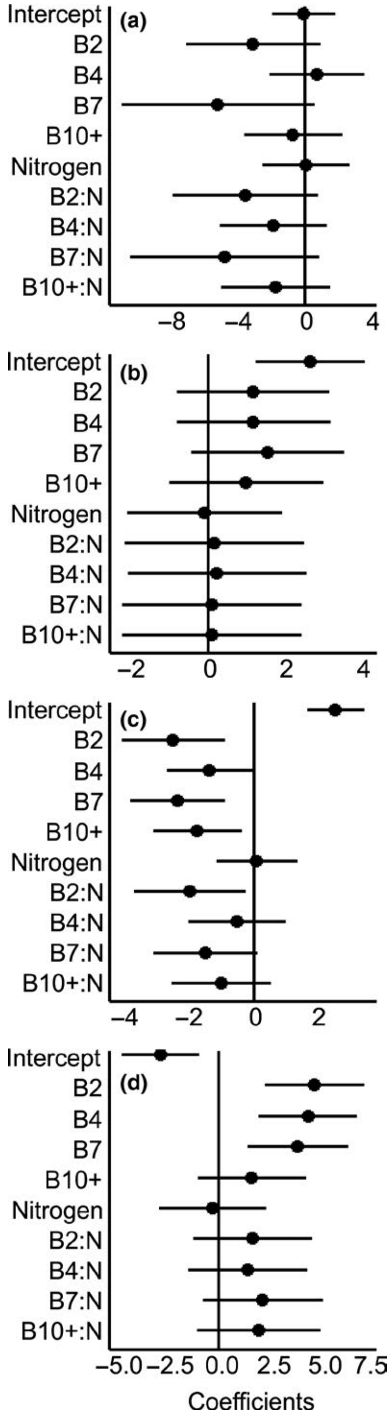

FIGURE 6 Coefficient plots of effects of moorland burning on the ecohydrology of river basins (EMBER) generalised linear mixed models (GLMMs) showing standardised coefficient estimates and 95% confidence intervals for the comparison of (a) *Sphagnum*, (b) *Calluna vulgaris*, (c) *Eriophorum vaginatum* and (d) *Campylopus introflexus* cover on B2, B4, B7 and B10+ plots (all n = 15) to unburned plots (n = 60); the relationship of cover with nitrogen deposition on unburned plots (nitrogen); and comparison of the relationship of cover with nitrogen deposition on B2, B4, B7 and B10+ plots to unburned plots (B2:N, *etc*.). Marginal R^2 = 0.54, 0.12, 0.57 and 0.51 respectively; Conditional R^2 = 0.72, 0.88, 0.79 and 0.70 respectively

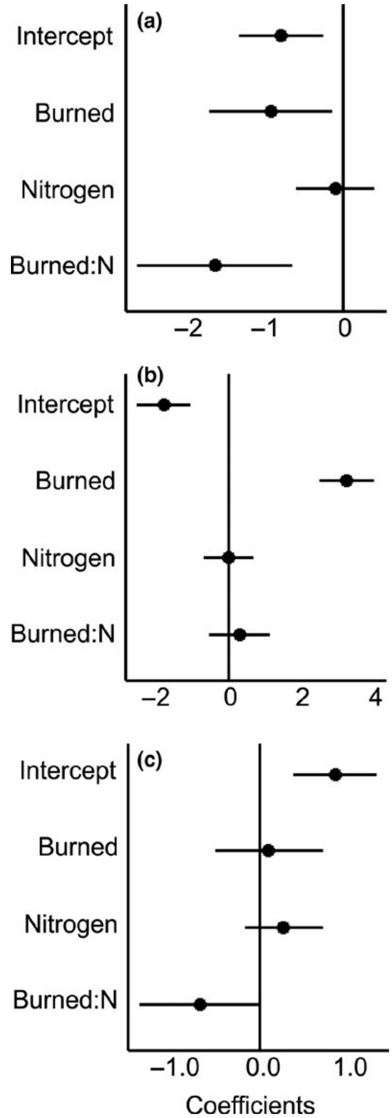

FIGURE 7 Coefficient plots of condition monitoring (CM) generalised linear mixed models (GLMMs) showing standardised coefficient estimates and 95% confidence intervals for the comparison of (a) *Sphagnum*, (b) *Calluna vulgaris* and (c) *Eriophorum vaginatum* cover on burned plots (n = 788) to unburned plots (n = 1,105); the relationship of cover with nitrogen deposition on unburned plots (Nitrogen); and comparison of the relationship of cover with nitrogen deposition on burned plots to unburned plots (Burned:N). Marginal R^2 = 0.09, 0.17 and 0.01 respectively; Conditional R^2=0.39, 0.71 and 0.49 respectively

can take several years (e.g. Brown et al., 2015; Holden et al., 2015). The vegetation of the unburned sites shows a clear divide between North and South Pennine sites in the EMBER NMDS ordination (Figure 2) in line with the two mire NVC types they supported (Table 1). However, this north–south divide is not apparent in the burned plots, and three of the five burned sites supported heath vegetation types associated with burning, grazing and atmospheric pollution (Elkington et al., 2001). This suggests that geographically variable vegetation community characteristics can be overridden by the effects of burning.

Although there was less clear separation of burned and unburned plots in the CM NMDS, clustering was still evident, indicating that many burned sites have similarities in vegetation composition on a national scale. The overlap between burned and unburned plots in the ordination space is unsurprising considering the range of burn ages and severities represented in the burned plots, and the overall range of species compositions in the dataset. Additionally, the CM NMDS was based on cover of 26 indicator taxa (Table S1). These taxa represent the majority of cover and the most ecologically important species on UK blanket peatland sites (Averis et al., 2004); however, it is possible that further differences in the vegetation composition of plots would be observed with the inclusion of rarer species.

4.2 | Variation in *Sphagnum* cover

Analysis of the CM data, compiled from 1893 plots at 85 sites across England, indicated lower *Sphagnum* cover on average where burning was identified. This finding is contrary to observations of *Sphagnum* cover at a single experimental burn site in northern England (Hard Hill, Moor House NNR) where *Sphagnum* cover was greatest on plots burned every 10 years (Lee et al., 2013). A reduced *Sphagnum* propagule bank was observed on burned plots at Hard Hill (Lee, Alday, Rose, O'Reilly, & Marrs, 2013), but this may have had a limited effect on cover because the experimental plots are surrounded by intact peatland vegetation which could have contributed to *Sphagnum* recruitment within the burned plots. In contrast, the CM sites represent a range of intact and degraded vegetation types and management strategies. Our results suggest that current burning practices may leave *Sphagnum* vulnerable to decline in some cases. This could reflect fire damage to *Sphagnum* similar to that described by Lindsay and Ross (1994), competitive effects or an indirect impact via peat physical and hydrological properties (Brown et al., 2015; Clay, Worrall, Clark, & Fraser, 2009; Holden et al., 2014, 2015).

Data from the 10 EMBER catchments showed that differences in *Sphagnum* cover between unburned plots and burned plots of any age were not significant, although effect directions were negative with the exception of B4 plots. These findings contrast with the significant relationship in the larger CM dataset. It is possible that changes in *Sphagnum* abundance after burning were asynchronous across sites, or that the number of EMBER plots (3 per burn age × 5 sites) did not control adequately for other factors affecting *Sphagnum* abundance (e. g. burn severity, microtopography and livestock access).

Nitrogen deposition had a negative relationship with *Sphagnum* cover in both datasets. Both nitrogen and sulphur (deposition of which were correlated) can have direct physiological impacts on *Sphagnum* (Ferguson et al., 1978; Granath et al., 2012), and nitrogen can increase competition from vascular plants (Limpens et al., 2011), promoting graminoid cover (Field et al., 2014). Sulphur deposition has declined faster than nitrogen deposition in recent decades (Curtis & Simpson, 2014), but legacy impacts on vegetation are possible.

The significantly more negative relationship between nitrogen deposition and *Sphagnum* cover on burned plots compared to unburned plots in the CM data may indicate that atmospheric pollution affects *Sphagnum* re-establishment by limiting growth or propagule availability after burning. *Sphagnum* peatlands are vulnerable to N deposition (Granath, Limpens, Posch, Mücher, & de Vries, 2014), and this interaction suggests that vulnerability may intensify if burning continues to increase. The positive relationship between *Sphagnum* and northing is difficult to attribute to a single influence, but may be associated with geographic variation in rainfall (Nijp et al., 2014), temperature (and evapotranspiration), geology or land management. The lower *Sphagnum* cover on plots with livestock droppings may be due to nutrient inputs, physical damage or hydrological impacts of trampling such as peat compaction leading to less water availability (Meyles, Williams, Ternan, Anderson, & Dowd, 2006). Finally, the relationship between elevation and *Sphagnum* was positive in the EMBER dataset and negative in the CM dataset. This difference may be a result of the larger range of elevations in the CM dataset (Table S2), or regional variation in the effect of elevation according to factors such as climate or abundance of competitors such as *C. vulgaris*.

4.3 | Variation in *Calluna vulgaris* cover

The greater *C. vulgaris* cover on burned sites in the CM data was expected, as burning is often practised to regenerate heather. It could be suggested that dwarf-shrub dominated vegetation is more likely to be selected for burning, but all plots were on deep peat (>30 cm), which suggests that this vegetation type is itself a legacy of management. On intact, *Sphagnum*-dominated blanket peatland, *C. vulgaris* is thought to regenerate naturally through layering of branches as *Sphagnum* grows up through the stems (Forrest, 1971; Macdonald, Kirkpatrick, Hester, & Sydes, 1995). Burning may disrupt this process and enhance *C. vulgaris* regeneration from seeds to roots (Forrest & Smith, 1975). The processes behind the increase in *C. vulgaris* may also be hydrological, as burning can result in deeper water-tables and less water availability (Holden et al., 2015), and *C. vulgaris* is primarily a heath species which can tolerate drier conditions than many other peatland plants.

The CM data also indicated a positive correlation between *C. vulgaris* cover and both elevation and northing value, which could be due to geographic variation of multiple influences as discussed for *Sphagnum*. Plots with livestock droppings had less *C. vulgaris* cover than those without, perhaps indicating a negative impact of grazing in accordance with the findings of Hulme et al. (2002). However, it is also possible that dense heather limits livestock access or other areas are grazed preferentially.

4.4 | Variation in *Eriophorum vaginatum* cover

The lower cover of *E. vaginatum* on plots burned 2, 7 and 10+years ago compared to plots at unburned EMBER sites was unexpected given the dominance after fire observed in past studies (Hobbs, 1984). However, in the CM data, there was no significant difference in *E. vaginatum* cover between burned and unburned plots. The ability of *E. vaginatum* to survive burning may depend on fire intensity, which was not quantified in this study, and the extent to which fire penetrates tussocks, damaging growing buds. Rapid regeneration of *C. vulgaris* after fire may also limit the opportunity for *E. vaginatum* to proliferate.

The positive relationship between *E. vaginatum* and both northing and elevation in the CM data may indicate enhanced competitive ability in cooler, wetter conditions. The interactive relationship of burning and nitrogen deposition with *E. vaginatum* observed in the EMBER data could indicate that after damage by burning, *E. vaginatum* is more susceptible to nutrient-driven competition (Wein & Bliss, 1973).

4.5 | Variation in *Campylopus introflexus* cover

Results from the EMBER data suggested that *C. introflexus* colonises rapidly after burning and can sustain increased cover for at least 7 years post burn. This is consistent with reports of *C. introflexus* colonising bare and disturbed peat (Equihua & Usher, 1993) and suggests that disturbance including burning may have assisted the spread of this non-native species in Europe. Cover was less at higher elevations, which could indicate climatic preferences or limited dispersal. Nitrogen deposition had a positive relationship with *C. introflexus*, consistent with reports of a positive response of the species to nitrogen (Southon et al., 2012), which suggests it may be able to occupy a niche vacated by pollution sensitive native species such as *Sphagnum*.

5 | CONCLUSIONS

Our results suggest that burning, atmospheric pollution and livestock presence are all associated with modified peatland vegetation. If the trend of further encroachment of prescribed burning onto blanket peatland continues, our results suggest that a shift towards greater cover of *C. vulgaris* and *C. introflexus*, less of the peat formers *E. vaginatum* and *Sphagnum*, and greater vulnerability to nitrogen impacts are possible. Livestock presence was also associated negatively with *Sphagnum* cover and we suggest that the use of burning and grazing as management tools on peatlands should be approached with caution where restoration or maintenance of active, peat-forming vegetation is an aim.

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AUTHORS' CONTRIBUTIONS

L.B., J.H. and S.P. were investigators on the EMBER project. D.J.G., A.C., J.H., S.P. and A.N. conceived the ideas for this paper. A.N. analysed the data and led the writing of the manuscript. All authors contributed to methodology and interpretation, critically assessed the drafts and gave final approval.

DATA ACCESSIBILITY

The data used for the analyses in this paper are archived in the Research Leeds Data Repository. DOI: https://doi.org/10.5518/230 (Noble et al., 2017).

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