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ORIGINAL PAPER

Peatland vegetation change and establishment of re-introduced *Sphagnum* moss after prescribed burning

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Abstract

Fire, including prescribed burning, is common on peatlands globally and can affect vegetation, including peat-forming *Sphagnum* mosses, and affect ecosystem services. We monitored vegetation in different burn-age categories at three UK peatland sites over a 19-month period. Half of the plots had *Sphagnum* fragments added and their survival was assessed. Changes in vegetation composition over time, and associations between vegetation composition, site and burn-age category were investigated. Plots in the most recently burned category were likely to have more bare peat, a thinner moss layer and lower vascular plant strata. Graminoid cover initially increased after burning but was low after 10+ years. Dwarf shrub cover increased after burning and remained high after 10+ years. At the most *Sphagnum*-rich site, a high proportion of existing *Sphagnum* cover was bleached one year after burning, but recovery occurred during the study period. *Sphagnum* re-introduction success decreased over the study period in the most recent and intermediate burn-age categories at the most *Sphagnum*-poor site. These results show that burning rotation length is an important factor in determining site-level vegetation composition on burned sites. More frequent burning will result in a greater proportion of land in the early post-burning stages, potentially resulting in a thinner moss layer, more bare peat and less healthy *Sphagnum*, with potential consequences for carbon balance. No evidence was found to support the use of burning as a tool to increase existing *Sphagnum* or promote *Sphagnum* re-establishment success.

Keywords Blanket bog · Fire · Heather · Land management · Moss · Succession

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Introduction

Peatlands are important carbon stores (Yu et al. 2010), covering around 423 million ha globally (Xu et al. 2018). In a healthy state, peatland ecosystems have the potential to accumulate carbon (Clymo et al. 1998), support biodiversity (Littlewood et al. 2010) and regulate hydrological processes (Labadz et al. 2010). However, degradation due to anthropogenic influences can threaten peatland function (Evans et al. 2014). In particular, vegetation change has the potential to harm the provision of ecosystem services (Grayson et al. 2010; Holden et al. 2008; Nichols et al. 2014; Ritson et al. 2016).

On northern hemisphere blanket peatlands, vegetation often largely consists of mosses, graminoids and dwarf shrubs. Vegetation composition plays a central role in ecosystem function, and although a range of species occur naturally on blanket peatlands, some may have a detrimental impact when dominant. Mosses can make up a significant component of the vegetation and include *Sphagnum* mosses which are of central importance for peat formation (van Breemen 1995) and water quality (Armstrong et al. 2012; Ritson et al. 2016), but require relatively wet conditions (Price and Whitehead 2001) and can be pollution-sensitive (Ferguson et al. 1978; Gunnarsson and Rydin 2000). A variety of pleurocarpous and acrocarpous moss species are also found on peatlands. However, differences in water storage capacity (Elumeeva et al. 2011) as well as net ecosystem exchange and decomposition rates (Orwin and Ostle 2012) mean that these groups do not support peat accumulation to the same extent as *Sphagnum*. Vascular plants including graminoids and dwarf shrubs contribute to the structural diversity of peatland habitats (Malmer et al. 1994) and can provide food and shelter for livestock and wildlife (Garnett et al. 2000; Robertson et al. 2017; Thompson et al. 1995). Some graminoid species, such as the sedge *Eriophorum vaginatum*, can also contribute to peat formation (Kalnina et al. 2015; McClymont et al. 2011). However, dominance of some vascular plants, such as *Calluna vulgaris*, may increase the amount of carbon lost as dissolved organic carbon (DOC) and CO_2, potentially owing to their impact on water table and soil temperature (Armstrong et al. 2012; Dixon et al. 2015).

Prescribed vegetation burning occurs on many peatlands worldwide for purposes including agricultural production and wildfire risk management. In the UK, it is often used as a vegetation management tool on blanket peatlands, particularly to improve habitat for the game bird red grouse (*Lagopus lagopus scotica*) on sport shooting estates (Douglas et al. 2015). Patches of up to around 4000 m^2 are burned on a rotation of c.7–25 years to produce a mosaic of vegetation ages suitable for foraging and nesting (Thacker et al. 2015). Burn severity varies, but good practice guidelines recommend 'cool' burns which remove the canopy layer of vegetation without consuming the moss or litter layer or igniting underlying peat (Defra 2007; Scottish Government 2011). The area of UK peatlands managed by burning has increased in recent decades (Douglas et al. 2015; Thacker et al. 2015; Yallop et al. 2006), causing concern about potential impacts on peatland function and debate about the sustainability of current practice (Allen et al. 2016; Brown et al. 2016; Harper et al. 2018).

Past research has shown that burning is associated with altered vegetation communities and that the abundance of key species can fluctuate in the years following burning (Noble et al. 2018). However, timescales of change after burning, and variation in the process between sites, remain poorly understood. For *Sphagnum*, understanding post-burning change is complicated by the fact that water deficiency or temperature damage can cause visible bleaching and reduced photosynthetic efficiency (Harris 2008; Taylor et al. 2017), but recovery of affected tissue or regeneration via new growth from this state may be possible (Taylor et al. 2017). Knowledge of when important vegetation changes occur in the post-burning trajectory will aid our ability to place past research into context and help to inform monitoring and land management decisions.

Sphagnum re-introduction via deliberate human intervention can be used as a tool to reinstate peat-forming vegetation as part of peatland restoration projects (Chirino et al. 2006; Ferland and Rochefort 1997; Gunnarsson and Söderström 2007; Robroek et al. 2009). *Sphagnum* re-introductions have taken place on some degraded UK peatlands, where historically abundant *Sphagnum* has been reduced by drivers including pollution and drainage (Carroll et al. 2009). *Sphagnum* can be added in the form of vegetative diaspores (Campeau and Rochefort 1996), plugs, or contained within manufactured products including beads and gel (Hinde et al. 2010). Past work has investigated influences on *Sphagnum* establishment success, including water availability (Noble et al. 2017; Robroek et al. 2009), nutrient status (Noble et al. 2017), propagule size (Gunnarsson and Söderström 2007) and climatic conditions (Chirino et al. 2006). However, it is not known how time since burning impacts establishment success of added *Sphagnum*. Increased knowledge in this area would be helpful when planning re-introductions on previously burned sites. Additionally, it has been suggested that burning may be useful as a restoration tool to remove dominant dwarf shrub canopies and facilitate *Sphagnum* establishment (Uplands Management Group 2017), but more evidence is needed to determine the effectiveness of such an approach (Lunt et al. 2011; IUCN 2017).

In this study, we aim to understand timescales of vegetation change and the potential for establishment of re-introduced *Sphagnum* after burning, and consider variation in these processes between sites. Specifically, we hypothesise that burning affects key plant groups with the capacity to influence ecosystem services. We also hypothesise that impact timescales differ between plant groups because of differences in their environmental tolerances and growth forms. By monitoring existing vegetation over 2 years at three blanket peatland sites, using plots which had different durations since they were last burned, we were able to construct a timeline of post-burning change in key plant groups and infer potential impacts on peatland ecosystem services. Measuring survival and growth of *Sphagnum* added to duplicate plots enabled assessment of how re-introduction for peatland restoration might interact with current burning regimes and burning undertaken for restoration purposes.

Methods

Three blanket peatland sites were chosen to represent key regions for this habitat in England; the Cheviot Hills, the North Pennine Hills, and the Peak District (Table 1). All three sites were managed for grouse shooting and had been burned on a rotational basis regularly for at least several decades. Their vegetation principally consisted of dwarf shrubs (mainly *Calluna vulgaris*), graminoids (including *Eriophorum vaginatum* and *Eriophorum angustifolium*) and mosses (a mix of pleurocarps, acrocarps and *Sphagnum* spp.), which occur in varying quantities at each site (Table 1).

At each site, five burn patches in each of three age categories were chosen with the help of land managers. Assessment of burn-age was based on site specific knowledge and morphology of *C. vulgaris*. After burning, *C. vulgaris* regenerates mainly from root stock and produces distinct new shoots during each year's growth season, which can be counted, enabling approximation of years since fire (Gimingham 1989). The youngest burn-age category (B1) comprised patches of vegetation burned approximately 12–18 months before the

Table 1 Site locations, mean living cover of key vegetation groups (taken from summer 2016 surveys; ± standard error) and modelled values for contemporary (2014–2016) atmospheric acid deposition (APIS 2018)

Site	Lat	Long	Dwarf shrub cover (%)	Graminoid cover (%)	Moss cover (of which *Sphagnum*) (%)	Acid (keq ha^{-1} $year^{-1}$)
Cheviot	55.455	− 2.112	52 ± 10	19 ± 6	35 ± 7 (3 ± 2)	1.41
North Pennine	54.864	− 2.396	37 ± 7	54 ± 6	77 ± 4 (29 ± 3)	1.51
Peak district	53.233	− 1.980	45 ± 8	28 ± 5	69 ± 4 (< 1 ± < 1)	1.98

All sites are on deep peat (> 40 cm) and managed for grouse shooting

first survey in March 2016. The intermediate burn-age category (B5) comprised patches burned approximately 5 years before the first survey, and the oldest category (B10 +) comprised patches burned at least 10 years before the first survey.

Within each patch, two 1 × 1 m plots were located 5 m apart and marked using bamboo canes. This gave a total of 30 plots per site; 5 pairs in each burn-age category. These plots were surveyed either four (Cheviot) or five (North Pennine and Peak District) times between March 2016 and October 2017. On each occasion, vegetation height and moss carpet thickness were measured at five points to 1 cm accuracy and the mean was recorded. Area cover of dwarf shrubs, graminoids, individual moss species, lichen, fungi, liverworts and other plants were recorded as percentages, as was cover of bare peat. For plants other than *Sphagnum*, only living cover was analysed. However, for *Sphagnum* both ‘healthy’ cover, and total cover including bleached patches, were analysed. Five plots (two pairs of B10 + plots and one B5 plot) were burned during the study at the Cheviot site and these were completely excluded from the analyses.

In July 2016, *Sphagnum* vegetative material was added to one of each pair of plots (a total of 45 plots over three sites). A mix of *Sphagnum* species representative of those present (comprising mainly *S. capillifolium*, with *S. papillosum* at the North Pennine and Cheviot sites and *Sphagnum subnitens* at the Peak District site) was collected at each site, and 10 ‘plugs’ of approximately 4 cm diameter and 10 cm length, consisting of several *Sphagnum* stems with the capitula facing upwards were added to each plot. These were inserted into the existing moss layer or surrounded by the vascular plants present. The survival (‘re-introduction success’) of the added *Sphagnum* was assessed during three subsequent surveys on a semi-quantitative percentage scale, where scores ranged from 0, indicating that all added *Sphagnum* was dead, dried out or absent, to 100, indicating that all added *Sphagnum* appeared alive and healthy.

All statistical analyses were carried out using R 3.1.0 (R Development Core Team 2010) and the packages vegan (Oksanen et al. 2013), nlme (Pinheiro et al. 2016), lsmeans (Lenth 2016) and ggplot2 (Wickham 2009). Firstly, an exploratory Non-metric Multidimensional Scaling (NMDS) ordination was carried out on the multivariate vegetation data from all surveys on the plots without *Sphagnum* additions. This enabled visual inspection of how plots from different sites and burn-age categories differed in their vegetation composition by projecting them into a two-dimensional ordination space where plots with less similar vegetation are located further apart.

Linear mixed models were used to determine how vegetation characteristics (vegetation height, moss depth, bare ground cover), cover of plant taxonomic groups (dwarf shrubs, graminoids, pleurocarpous mosses, acrocarpous mosses, *Sphagnum* spp.) and *Sphagnum* re-introduction success varied according to site, burn-age category, time during the study, and the interactions between these variables. Aggregating individual moss species into groups enabled investigation of how groups of related species responded to predictor variables, including species which were too infrequent to analyse independently or absent from one or more sites. Dependent variables recorded as percentages were logit transformed for this analysis to satisfy model assumptions and improve fit, assessed by visual inspection of residual plots. Time during the study was treated as a continuous variable and was centred so that the mid-point of the study acted as the reference value. Plot ID was included in each model as a random effect to account for the repeat surveys. Least-squares means were compared (with Tukey's adjustment for multiple comparisons) to reveal significant differences between site and burn-age combinations. All differences described in the results are significant at $p < 0.05$.

Results

The NMDS ordination results (Stress = 0.184, K = 2) suggest that vegetation communities differed according to both site and burn-age category. The North Pennine site was associated with the positive region of axis 1, whereas the Peak District and Cheviot sites were more associated with the negative region (Fig. 1a). Meanwhile, the most recently burned (B1) plots were more likely to fall in the positive region of axis 2, with the least recently burned plots (B10+) falling in the negative region, and the intermediate (B5) plots were spread around the origin in both positive and negative regions (Fig. 1b).

Individual vegetation characteristics and taxonomic groups showed differences according to site and burn-age, and change over the course of the study for some variables (Online Resource 1). Bare peat was greater in B1 plots than B10+ plots at the Peak District site and greater in B1 plots than B5 and B10+ plots at the Cheviot site, with a decrease over time in the B1 plots (Fig. 2a). Moss depth was greater in B10+ plots than B1 plots at both the North Pennine and Peak District sites (Fig. 2b). Meanwhile, vegetation height was greater in B10+ plots than B1 plots at the Peak District and Cheviot sites and decreased over time in B1 plots at the North Pennine site (Fig. 2c).

Dwarf shrub cover increased over the study period in B1 plots at the Cheviot site and B1 and B5 plots at the North Pennine site. At all three sites B10+ plots had greater cover than B5 and B1 plots, and at the Cheviot site B5 plots also had greater cover than B1 plots (Fig. 3a). Graminoid cover increased over the study period in B1 plots at all three sites, but was greater in B5 plots than B10+ plots at the Cheviot site, greater in B1 plots than B10+ plots at the North Pennine site, and greater in both B1 and B5 plots than in B10+ plots at the Peak District site (Fig. 3b).

Pleurocarpous moss cover was greater at the Peak District site than the Cheviot site, and increased over the study period in Cheviot B1 plots, but showed no significant differences according to burn-age (Fig. 4a). Acrocarpous mosses increased over the study period in B1 plots at the Cheviot site (Fig. 4b). For existing *Sphagnum*, healthy cover increased over time in B1 plots at the North Pennine site (Fig. 5a), while total cover (including bleached areas) decreased over time in B1 plots at the Cheviot site (Fig. 5b). Cover of both healthy and total existing *Sphagnum* was greater at the North Pennine site than the other two sites (Fig. 5).

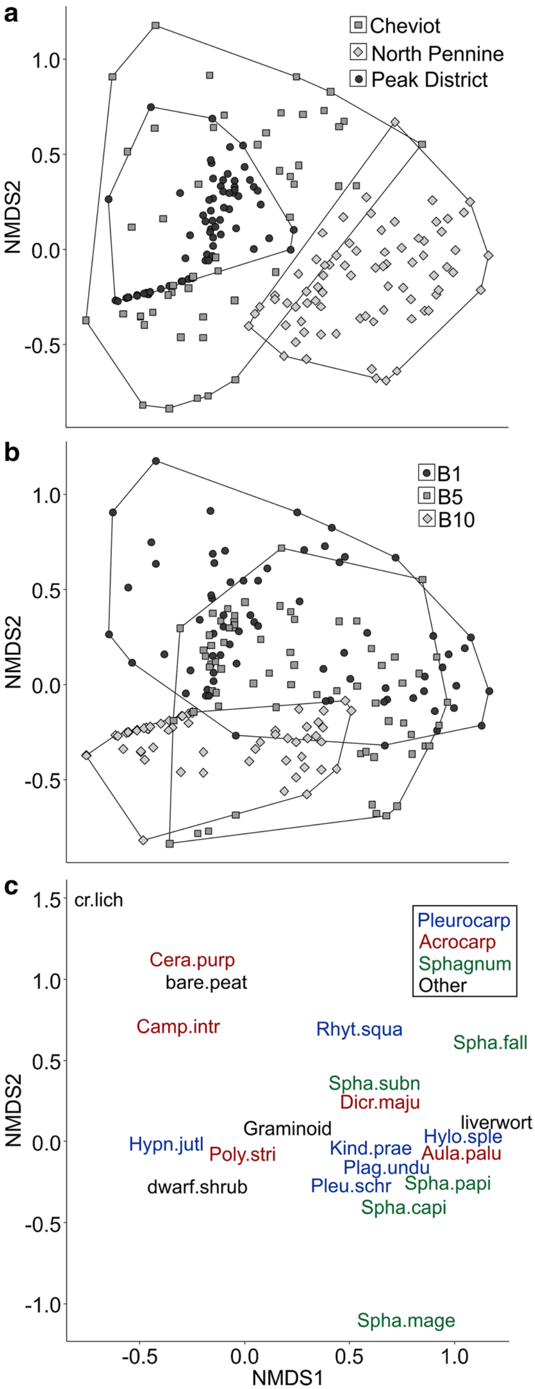

Fig. 1 NMDS ordination of vegetation data from 201 blanket peatland plots showing **a** dissimilarities between plots according to site, **b** dissimilarities between plots according to burn-age (B1—burned 1 year before start of study, B5—burned c.5 years before start of study, B10—burned at least 10 years before start of study) and **c** species associations (*Aula.palu = Aulacomnuim palustre, Camp.intr = Campylopus introflexus, Cera.purp = Ceratodon purpureus, Dicr.maju = Dicranum majus, Hylo.sple = Hylocomium splendens, Hypn.jutl = Hypnum jutlandicum, Kind.prae = Kindbergia praelonga, Plag.undu = Plagiothecium undulatum, Pleu.schr = Pleurozium scherberi, Poly.stri = Polytrichum strictum, Rhyt.squa = Rhytidiadelphus squarrosus, Spha.capi = Sphagnum capillifolium, Spha.fall = Sphagnum fallax, Spha.mage = Sphagnum magellanicum, Spha.papi = Sphagnum papillosum, Spha.subn = Sphagnum subnitens*)

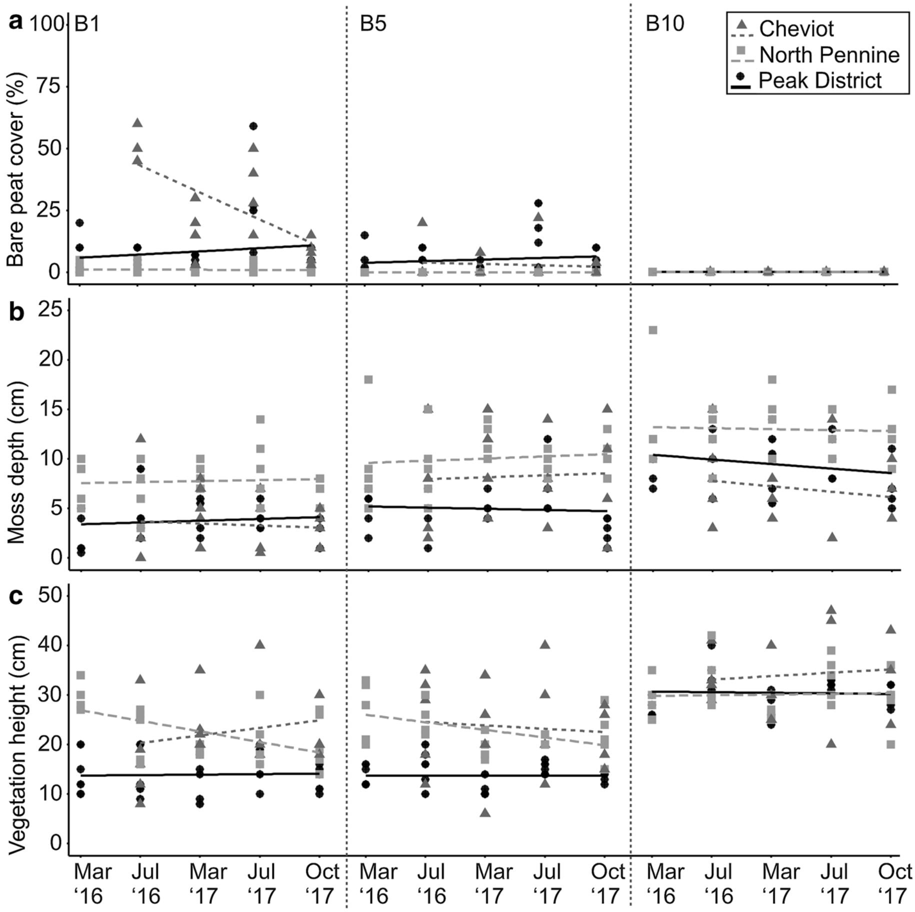

Fig. 2 Change over time with fitted relationships for **a** bare peat, **b** moss depth and **c** vegetation height according to site and burn-age category (B1—burned 1 year before start of study, B5—burned c.5 years before start of study, B10—burned at least 10 years before start of study)

In plots where *Sphagnum* was added, there was no significant difference in re-introduction success according to burn-age. However, there was a decrease in success score over the recording period in B1 and B5 plots at the Peak District site. Success was greater at the North Pennine site than the Peak District site within the B1 category, and greater at the North Pennine site than the Cheviot site in the B10 + category (Fig. 6).

Discussion

The NMDS ordination suggested that time since burning is an important factor in determining vegetation composition, alongside local site conditions. The B10 + plots appeared to occupy a smaller area of the NMDS ordination than B1 or B5 plots, indicating greater similarity between plots, which could be the result of a few species or groups (such as dwarf shrubs) common to all three sites dominating at this stage of the burning cycle.

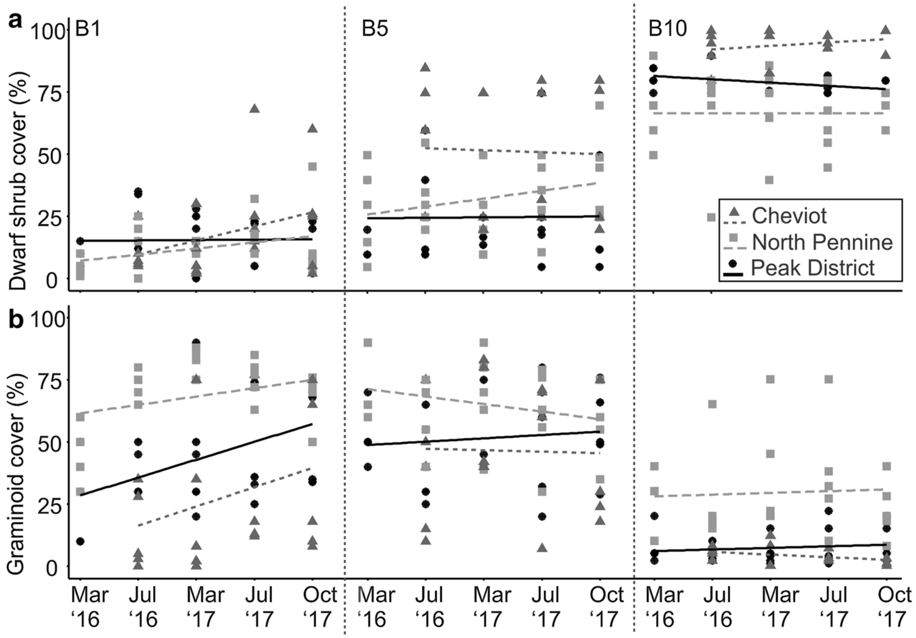

Fig. 3 Change over time with fitted relationships for **a** dwarf shrub cover and **b** graminoid cover according to site and burn-age category (B1—burned 1 year before start of study, B5—burned c.5 years before start of study, B10—burned at least 10 years before start of study)

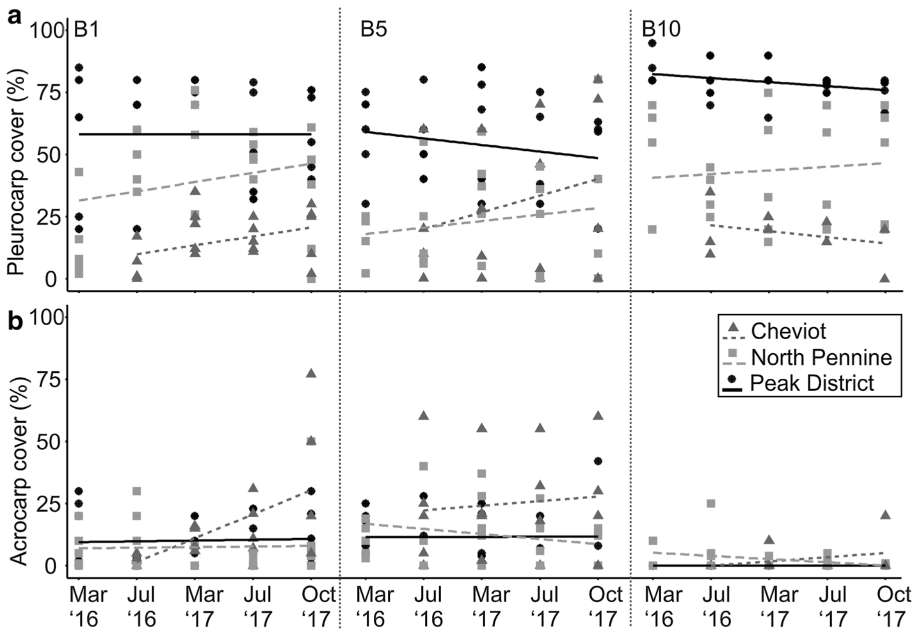

Fig. 4 Change over time with fitted relationships for **a** pleurocarpous moss cover and **b** acrocarpous moss cover according to site and burn-age category (B1—burned 1 year before start of study, B5—burned c.5 years before start of study, B10—burned at least 10 years before start of study)

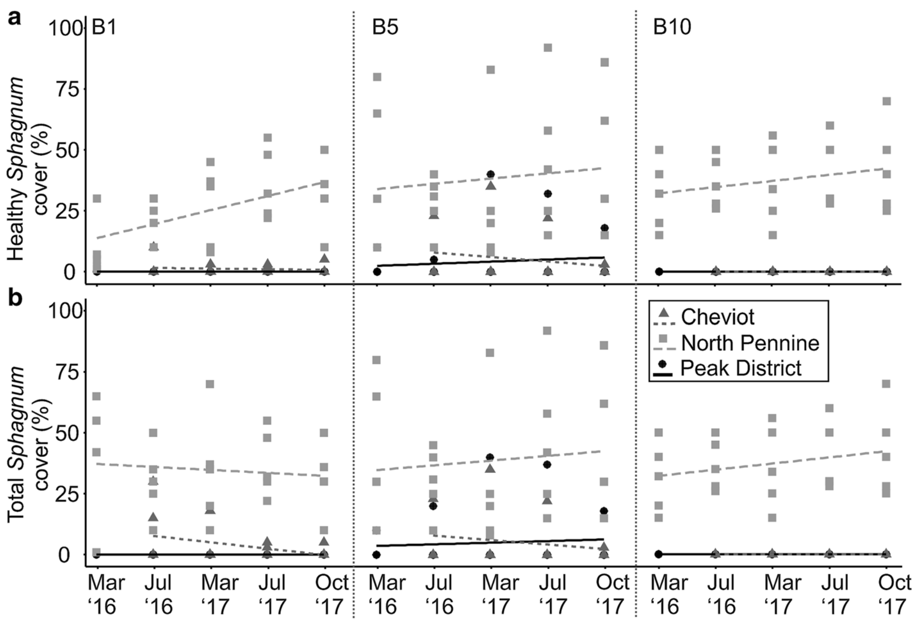

Fig. 5 Change over time with fitted relationships for **a** healthy Sphagnum cover and **b** total Sphagnum cover (including bleached patches) according to site and burn-age category (B1—burned 1 year before start of study, B5—burned c.5 years before start of study, B10—burned at least 10 years before start of study)

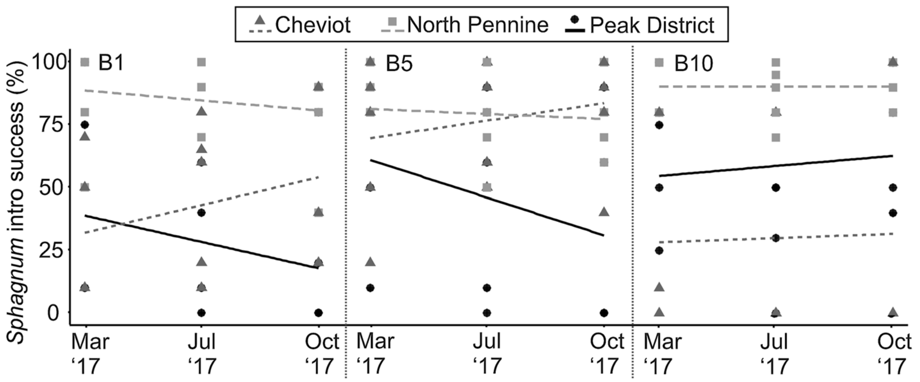

Fig. 6 Change over time with fitted relationships for Sphagnum re-introduction success according to site and burn-age category (B1—burned 1 year before start of study, B5—burned c.5 years before start of study, B10—burned at least 10 years before start of study)

The results of the univariate analyses suggest that burning can increase bare peat and cause thinning of the moss layer, probably due to moss being consumed or desiccated by fire. This is consistent with the reduced bryophyte biomass observed in more recently burned plots by Ward et al. (2007). Such changes could potentially result in faster overland flow (Holden et al. 2008), increased vulnerability to erosion (Holden et al. 2007), increased DOC in water courses (Yallop et al. 2010), and decreased carbon storage in the moss layer (Ward et al. 2007) in the years following burning. Bare peat was absent in the B10+plots, indicating that revegetation occurred within the rotational burning cycle at these sites. This suggests that effects of bare peat such as faster overland flow (Holden et al. 2008) are likely to be temporary. However, the extent of such effects will depend on the overall proportion of recently burned ground, which is dictated by the burning rotation length.

The lower vegetation height observed in the most recently burned plots at two out of three sites was expected owing to loss of canopy layer dwarf shrubs during fire. The decrease in vegetation height over time in B1 plots at the North Pennine site could be due to partially burned dwarf shrubs remaining after the burn before gradually reducing owing to mechanical damage from weather or trampling. The residual burned stems could indicate that burn severity was lower at the North Pennine site, a theory which is supported by the lower proportion of bare peat observed after burning compared to the other sites. Differences in burn severity can be a function of fuel structure and moisture (Davies and Legg 2011), wind speed (Santana and Marrs 2014) and burning technique (e.g. fuel assistance, fire direction relative to wind and patch size).

The peak in graminoid cover in the early years after burning, along with the continued increase in dwarf shrub cover suggest that burning on a shorter rotation may lead to graminoid dominance, while longer rotations are likely to favour greater dwarf shrub cover. This follows a similar pattern to past observations of burning effects on these groups (Harper et al. 2018; Lee et al. 2013; Ward et al. 2007). Past research suggests that fluctuations in the abundance of vascular plant groups are likely to lead to fluctuations in the export of DOC (Armstrong et al. 2012). The dwarf shrub dominance observed 10+ years after burning could potentially lead to increased DOC in watercourses draining the catchment, with implications for the cost of water treatment where catchments supply potable water (Dixon et al. 2015).

Analysis of the three moss groups indicated that pleurocarpous mosses persist throughout the burning cycle. This could suggest fire tolerance in some pleurocarpous species, as well as tolerance to changes in light, moisture availability and soil water chemistry (Brown et al. 2014) during the burning cycle. Acrocarpous mosses increased over the study period in B1 plots at the Cheviot site, possibly owing to the higher proportion of bare peat at this site after burning. *Ceratodon purpureus*, an acrocarpous moss common in the Cheviot B1 plots, has previously been observed to colonise bare ground after fire (Duncan and Dalton 1982; Thomas et al. 1994), as have other acrocarpous mosses including *Campylopus introflexus* (Equihua and Usher 1993; Noble et al. 2018). It is possible that extensive colonisation by acrocarpous mosses after fire could negatively affect the regeneration of other vegetation owing to a tendency to rapidly carpet bare ground, providing a poor substrate for young plants of other species, and potentially competing for resources (Equihua and Usher 1993). However, the ecological consequences of such an effect require further research.

No significant impact of burning on *Sphagnum* was observed at the Peak District site, which had low overall abundance in all burn-age categories. At the North Pennine site where abundance was much higher, cover of healthy *Sphagnum* increased over time in the B1 plots. However, total *Sphagnum* cover (including bleached patches) remained constant in the same plots, suggesting that the increase in healthy *Sphagnum* represented the recovery of bleached patches which had been damaged as a result of fire. The temporary bleaching is likely to be associated with a reduction in photosynthetic efficiency (Taylor et al. 2017), potentially affecting growth rate and carbon balance during this period. In B1 plots at the Cheviot site, healthy *Sphagnum* cover remained constant, but total *Sphagnum* cover decreased over time, suggesting that some of the bleached patches did not recover and were lost. However, this trend over time was driven by relatively low cover in a few patches so caution is needed in interpreting this result. Differences in *Sphagnum* abundance (and vegetation composition more generally) between sites could be the result of variation in current and historic atmospheric pollution (Table 1), which is known to affect several peatland plants including *Sphagnum* (Noble et al. 2018). Other potential causes of differences between the sites include climate (e.g. temperature, rainfall, seasonality), management history (e.g. grazing, drainage, past fire) and other site specific factors (topography, peat physical properties). These factors may have contributed to the formation of different vegetation composition at the three sites and may have influenced *Sphagnum* abundance in particular via differences in water availability, nutrient availability and competition.

Sphagnum re-introduction success was consistently high at the North Pennine site, which also had the greatest existing *Sphagnum* abundance. This could indicate that *Sphagnum* abundance at the other two sites was limited by environmental factors, such as water availability, rather than propagule availability. At the Peak District site, *Sphagnum* re-introduction success declined over the study period in B1 and B5 plots. This effect could be due to *Sphagnum* plugs drying out over time, as these plots had less dwarf shrub cover and shorter vegetation so were more exposed, and potentially subject to hotter summer temperatures (Brown et al. 2015). These results do not support the theory that removing the dwarf shrub canopy benefits *Sphagnum* growth as suggested in current management guidance (Uplands Management Group 2017).

Conclusions

The results of this work suggest that different burning frequencies will result in different vegetation composition outcomes, with shorter rotations generally favouring graminoids and longer rotations favouring dwarf shrubs. Vegetation therefore has the potential to affect ecosystem services in different ways over the course of the burning cycle. The timescale and extent of vegetation change after burning varied between the three study sites, suggesting that some sites may be more resilient to burning. However, repeated burning, or stressors such as drainage, pollution, grazing or climate change could decrease resilience. Finally, this study found no evidence to suggest that burning is effective as a restoration tool to encourage *Sphagnum*, or that removing the canopy by burning increases *Sphagnum* re-introduction success.

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Data availability The dataset generated and analysed during this study will be made available in the Research Data Leeds repository, a DOI will be minted and provided in due course.

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