Immediate effects of heather cutting over blanket bog on depth and microtopography of the moss layer

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SUMMARY

The blanket bog ecosystems of the UK uplands fulfil important biodiversity and ecosystem services at multiple spatial scales. In response to perceived negative effects of burning to control heather growth on blanket bog, regulations on vegetation management now restrict this practice as part of the UK's wider commitment to reducing carbon emissions. Instead, there has been a significant shift towards cutting as an alternative heather management tool, despite a notable lack of research on its effects on blanket bog. A key component of blanket bog ecosystems in the UK uplands is the moss layer, in which *Sphagnum* mosses play a particularly significant role. To investigate the immediate effects of heather cutting on the structure and extent of the moss layer, measurements of moss depth and cover at the end of the cutting season were taken from cut and uncut areas of two managed blanket bog sites in northern England. Measurements of moss depth were used to generate an index of moss microtopography. Cutting resulted in significant reductions of moss depth, heterogeneity of moss hummock microtopography, and extent of pleurocarpous moss, but there was no change in cover of *Sphagnum* or acrocarpous mosses. Further research is needed to understand the effects of cutting, particularly in the longer term, in terms of cutting height, type of machinery used, whether brash is removed, and repeat cut interval. This research will inform evidence-based management of blanket bog habitats.

KEY WORDS *Calluna vulgaris*, heather management, *Sphagnum*

INTRODUCTION

The peatlands of the UK uplands are of national and international importance for both biodiversity and ecosystem services and locally have a major socioeconomic role (Van der Wal *et al.* 2011, Natural England 2015). These habitats support a range of rare, threatened and declining flora and fauna, including specialised species that are adapted to spending at least part of their life cycle in a waterlogged, mostly acidic, nutrient-poor environment (Littlewood *et al.* 2010). A particularly notable feature of more northerly UK peatlands is the ground-nesting bird assemblage including golden plover *Pluvialis apricaria* L. and dunlin *Calidris alpina* L. (Newton 2020). A significant proportion of UK peatland is blanket bog - a layer of ombrotrophic peat that, in England, is defined as any peat over 40 cm deep and, together with upland valley mires, comprises an area of 3553 km^2 (Natural England 2010). The acidic, waterlogged condition of blanket bog means decomposition rates are slow and organic matter accumulates to form peat, sequestering carbon. Conservation of peatland through restoration and maintenance of these conditions has been highlighted as key to achieving current carbon goals through reduced greenhouse gas emissions and increased carbon sequestration (Dunn *et al.* 2021). Attaining these goals will require retaining and restoring the moss layer. Together with grasses, forbs and dwarf shrubs, the moss layer forms a protective skin across the peat surface, preventing erosion and drying out. *Sphagnum* mosses are particularly important members of this surface flora (Rochefort 2000), helping to reduce water flow from the uplands through their ability to store water (Holden *et al*. 2008), contributing to water quality through associated reductions in dissolved organic carbon in peatland waters (Armstrong *et al*. 2012, Ritson *et al*. 2016), maintaining the low pH of the peat body, and being a key peat-forming group of plant species (van Breeman 1995). Heterogeneity in the surface topography also provides favourable habitat for ground-nesting birds, for which the hummock and hollow structure within a generally open landscape provides damp, invertebrate-rich feeding opportunities while still affording visibility of approaching predators (Newton 2020). The surface roughness also reduces water runoff, maintaining higher water tables and associated anoxic conditions (Aleina *et al*. 2015, Moore *et al*. 2019), and makes an important contribution to carbon storage (Wang *et al*. 2021).

In the UK, blanket bog is subject to environmental regulations aimed at maintaining its ecological function as a carbon store, as part of the Government's wider commitment to achieve net zero carbon emissions by 2050 (UK Government 2021). Until recently, prescribed burning has been a widely used vegetation management practice, particularly on grouse moors that typically have large areas of heather-dominated moorland. This management practice not only prevents natural succession of scrub and woodland, but also creates a patchwork mosaic of varying vegetation age, height and structure (Robertson *et al*. 2017). The extent of moorland under burning management has been shown to be positively correlated with abundance of breeding red grouse (*Lagopus lagopus scoticus* Latham), golden plover, curlew (*Numenius arquata* L.), whinchat (*Saxicola rubetra* L.) and lapwing (*Vanellus vanellus* L.) (Tharme *et al*. 2001). For all species, older, taller heather provides cover from predators, while younger, shorter heather provides more nutritious shoots for adult grouse to forage on and higher insect availability for chicks (Gardner & Usher 1989, Palmer & Bacon 2001, Buchanan *et al*. 2006).

Over the last few decades, heather burning rates have increased (Yallop *et al.* 2006, Douglas *et al.* 2015). Despite this increase, evidence from uplands known as the Peak District (England) showed that burning met target burn sizes and covered less than the recommended total burn area (Allen *et al.* 2016), giving confidence that these increases in burning were not contrary to guidance at the time (DEFRA 2007). However, the regulations for England were recently updated to restrict the burning of ling heather *Calluna vulgaris* (L.) Hull and other vegetation on blanket bog (DEFRA 2021). This policy change was a response to expanding research into effects of burning management, with some negative effects reported but also much uncertainty remaining. For example, some studies suggest that repeat burn intervals of up to ten years will promote *Sphagnum* growth (Milligan *et al.* 2018, Whitehead & Baines 2018, Whitehead *et al.* 2021), while work at larger scale has shown a negative relationship between *Sphagnum* cover and heather burning, although this may be influenced by other factors including grazing and atmospheric pollution (Noble *et al.* 2017). Harris *et al.* (2011) recommended regular burning to maintain the species diversity of upland moorland plant communities by preventing heather dominance, despite burning being the cause of this dominance. Holden *et al*. (2014) reported negative effects on blanket peat hydrology at a range of scales, although Ashby & Heinemeyer (2019) have questioned the experimental design and interpretation of findings, identifying a need for further research that controls for site-specific differences in responses. Studies of the effects of managed burning on dissolved organic carbon (DOC) and water colouration provide evidence for increasing (Yallop & Clutterbuck 2009, Clutterbuck & Yallop 2010, Yallop *et al*. 2010, Clay *et al*. 2012, Grayson *et al*. 2012, Ramchunder *et al*. 2013), decreasing (Worrall *et al*. 2007, Clay *et al*. 2009, Worrall *et al*. 2013) and unchanged (Ward *et al*. 2007, Clay *et al*. 2009, Chapman *et al*. 2010, Clay *et al*. 2010, 2012; Brown *et al*. 2013, Worrall *et al*. 2013) levels of both factors. Although distillation of all these findings leads to the overall conclusion that burning on moorlands is correlated with an increase in DOC and water colour, it also highlights that plot-scale and catchment-scale studies are likely to exhibit different responses to burning, and that results are further influenced by whether runoff water, soil water or stream water is sampled (Harper *et al.* 2018). Despite continued uncertainty regarding the effects of managed burning, tighter restrictions of heather burning over blanket bog were introduced to protect the moorland ecosystem and the services it provides.

In response to increased restrictions on burning, there has been a significant shift towards cutting as an alternative heather management tool, despite a paucity of evidence on its effects. Much of the available research is not based in areas of deep peat (Calvo *et al.* 2002, Härdtle *et al.* 2009), and evidence for impacts on deep peat habitat is restricted to a few studies. Worrall *et al.* (2013) found that both heather cutting and heather burning reduced levels of DOC in soil water through an increase in water table level. Heather cutting has also been shown to reduce peat surface microtopography, but not to reduce peat depth or increase bulk density through compaction of the peat (Heinemeyer *et al.* 2019). Effects on fauna are even less well documented. Invertebrate surveys of vegetation and at ground level on heather cuts showed both species richness and Simpson diversity increased and then decreased with time after cutting and gave a recommended rotation of 15–20 years (Sanderson *et al*. 2020).

As with heather burning, cutting heather on UK sites designated for blanket bog requires statutory consent, which is conditional on there being no damage to the blanket bog habitat. However, our own anecdotal observations of management cuts indicate that some of this cutting is having a physical impact on the moss layer. Although some guidance exists (MacDonald 1996), the increasing reliance on cutting as a management tool means that more research is needed to understand better the effects of cutting on blanket bog community composition and ecological functioning, including hydrology and carbon fluxes. The recent change in methods of vegetation management provides an opportunity to explore these effects and to consider whether this alternative approach will have the positive outcome that is intended.

This study contributes to the knowledge base on heather cutting to support informed, evidence-based management decisions. We focus on the moss layer owing to its important role in the ecosystem. Using plots that had been cut within the previous six months, we consider the short-term effects of cutting. The conclusions of Heinemeyer *et al.* (2019), that surface microtopography is reduced by slicing of the tops of sedge hummocks, raises the question of whether moss hummocks could be similarly damaged by the cutters. Here we test the hypothesis that heather cutting results in an immediate reduction in overall moss depth and the heterogeneity of moss hummock topography by comparing moss depth between recent (within the last six months) heather cuts and surrounding uncut vegetation on two sites. The data presented here will also form a baseline against which future longer-term responses to cutting may be monitored.

METHODS

Study areas

The study was conducted on two sites, both moors managed for driven red grouse shooting, in Upper Teesdale in northern England. The sites share a similar sub-arctic oceanic climate with Moor House Environmental Change Network (ECN) site, 14 km from Site A and 6 km from Site B, for which the most recent mean annual temperature is 5.9 ºC and mean annual rainfall is 2028 mm (http://data.ecn.ac.uk/sites/ecnsites.asp?site=T04). The total area of Site A is 4.8 km^2. The mean altitude of experimental plots here was 574 m (range 552–595 m) with a mean peat depth of 138 cm (range 65–220 cm). On Site B, which has a total area of 2.7 km^2, mean plot altitude was 508 m (range 498–517 m) and mean peat depth was 148 cm (range 70–350 cm). The sites support a mix of mire communities, including *Erica tetralix*-*Sphagnum papillosum* raised and blanket mire (National Vegetation Classification (NVC) M18) and *Calluna vulgaris*-*Eriophorum vaginatum* blanket mire (NVC M19) (Rodwell 1991). Both sites are sheep-grazed from spring to autumn at no more than 0.1 ewes ha^{-1}. On Site B, approximately 100 separate small plots (each within the range of 4.5–8 m wide and 8–12 m long) of mature heather had been cut in winter 2020/21 across 0.38 km^2 of blanket bog. On Site A, an area of blanket bog of the same size as that on Site B was identified which contained ~ 60 heather cuts, of similar size to those on Site B, that had also been conducted in winter 2020/21. Cuts on both sites were done with a Softrack all-terrain vehicle and flail mower forage harvester which gathered the cut material from the cut area and scattered it up to approximately 5 m over the surrounding vegetation.

Selection of study plots

Within the defined 0.38 km^2 of blanket bog on each site, we randomly generated ten grid references with a minimum spacing of 50 m using QGIS v3.6 (QGIS Development Team 2019). At each random point we visually selected on the ground a suitable cut with an immediately adjacent uncut control plot, each sharing similar gradient and aspect, as close as possible to the randomly generated grid reference. Cut plots were a minimum of 5 m × 8 m. Control plots were stands of mature heather at least 40 m^2 (usually 5 m × 8 m, but availability of suitable habitat sometimes dictated different dimensions) with no visible evidence of cutting or burning, no keeper records of management within the last ten years, not smothered by brash from the adjacent cut, and undamaged by cutting machinery tracks. If we could not find a suitable pair of plots within 50 m of the random grid reference, we discarded that point and used a further randomly generated point, ensuring that the 50 m minimum spacing between pairs of plots was still observed. This gave 10 randomly selected, spatially separated pairs of plots, each pair comprising one cut and one control plot, on each site (Figure 1).

Vegetation measurements

As cutting had already taken place before the start of the study, there was no opportunity to collect pre-treatment baseline data. We collected all vegetation measurements between 23 March 2021 and 21 April 2021, after the winter in which cutting took place. We collected all data within a plot on the same day, and measured its paired cut or control plot on the same or the next day. Although the experimental design was limited by the absence of pre-treatment baseline measurements, control plots were immediately adjacent to cut plots, in the same stand of heather, and were measured within the same season as the cutting had taken place, so it can reasonably be assumed that they were representative of pre-treatment conditions.

Figure 1. Paired cut and control treatment plots on Site A (top) and Site B (bottom).

Moss depth and microtopography sampling

Within each cut and control plot, we established four 5 m transects, with inter-transect spacing of 1 m (minimum) to 3 m (maximum), on which we measured moss depth at 0.5 m intervals, giving 44 measurements per plot. In cut plots, transects ran perpendicular to the cutting direction to avoid placement along machinery tracks. We took cutting direction to be the same as the direction of any lines of vegetation remaining in the plot or, if not present, following the longest dimension of the plot. At each sampling point, we measured moss depth to 1 cm accuracy using a nylon cable rod inserted into the moss layer until the peat surface was reached, and recorded each moss depth measurement as *Sphagnum* or non-*Sphagnum*.

Vegetation height and moss cover

We positioned five 1 m^2 quadrats in each plot, one 2 m from each plot corner and one in the centre. Owing to the small size of available cuts and control areas there were some plots (seven on Site A and six on Site B) in which we had to reduce spacing to 1 m. The shape of two control plots, one on each site, meant that we placed quadrats in a single line along the centre of the plot to maintain the minimum 1 m spacing. Quadrats did not overlap with transects, and we avoided trampling quadrats and transects during sampling.

Within each quadrat we estimated horizontal cover of *Sphagnum* moss, acrocarpous mosses, and pleurocarpous mosses to the nearest 5 %. If present but below 5 % cover, we recorded as a tick and gave a value of 1 % in the analyses. We speciated *Sphagnum* mosses and recorded them as present where identifiable although, in some cases, cutting had removed identifying features. In each quadrat, we took five vegetation height measurements, one at each quadrat corner and in the centre. We measured vegetation height by placing a wooden sward stick marked with 1 cm denominations vertically into the vegetation and recording the height of the tallest vegetation in contact with the stick, to the nearest centimetre. At this time of year, there were no tall flowering grass stems, measurements of which would have positively skewed the mean value of vegetation height.

Data analyses

We derived a plot-level index of moss microtopography from the standard deviation of the difference, and direction of that difference, in moss heights measured at adjacent points along each transect. We then calculated the mean of those transect-derived values to generate a plot-level index. We calculated plot means of all transect and quadrat-derived data and then natural log-transformed before analysis. Resulting standardised residuals were normally distributed when checked against fitted values in residual plots.

We used transect sampling data from Site B to calculate plot mean moss depth for each moss type (*Sphagnum* or non-*Sphagnum*), weighted by sample size. A similar analysis for Site A was not possible due to low sample sizes of *Sphagnum* moss. We performed an ANOVA on these data to test whether *Sphagnum* mosses were more affected than non-*Sphagnum* mosses by cutting. We included moss type and treatment nested within treatment plot pairs as explanatory variables, with moss depth as the response variable. This showed no interaction of moss type and treatment effect ($F_{21,18} = -1.60$, $P = 0.13$), so we combined moss types for all analyses considering moss depth.

We considered site and treatment effects for all transformed variables in a nested ANOVA for moss depth, moss microtopography index, vegetation height and percentage cover of *Sphagnum*, pleurocarpous and acrocarpous mosses. For moss microtopography there was a site*treatment interaction effect (Table 1) and so, for this variable, we then considered treatment in a separate paired t-test for each site. The analyses were carried out in Genstat (21st edition).

RESULTS

Effects of cutting

Moss depth, index of moss microtopography, vegetation height and percentage cover of pleurocarpous mosses were all significantly lower on cut plots than on control plots and, with the exception of index of moss microtopography, this effect was consistent between sites (Table 1). Overall, moss depth was almost 40 % lower on cut plots (Table 2, Figure 2) and vegetation height was 62 % less (Table 2). Mean percentage cover of pleurocarpous moss was on average 13 % lower on cut plots.

Cutting had a significant effect on moss microtopography at both sites, but the initial analyses also showed a site*treatment interaction (Table 1). Consideration of treatment differences for each site separately showed both to be significant (Site A: $t = -2.72$, $P = 0.02$; Site B: $t = -6.03$, $P < 0.001$), but these effects were significantly greater on Site B than on Site A (Figure 3), with reductions in index of 33 % on Site B versus 13 % on Site A (Table 2), resulting in a similar post-cutting index on both sites (Figure 3).

Table 1. Effect of site, treatment (cut or no-cut control) and site*treatment (nested ANOVA) on moss depth, index of moss microtopography, vegetation height and % cover of *Sphagnum*, pleurocarpous and acrocarpous mosses cover from ten pairs of control and treatment plots on each of two sites. Significant P values are in bold.

Variable	Site	Treatment	Site*Treatment
Moss depth (cm)	$F_{1,18}$=3.63, P=0.27	$F_{1,18}$=78.54, **P<0.001**	$F_{1,18}$=0.67, P=0.42
Index of moss microtopography	$F_{1,18}$=0.65, P=0.43	$F_{1,18}$=41.22, **P<0.001**	$F_{1,18}$=9.25, **P<0.01**
Vegetation height (cm)	$F_{1,18}$=57.37, **P<0.001**	$F_{1,18}$=218.7, **P<0.001**	$F_{1,18}$=2.76, P=0.11
Sphagnum moss % cover	$F_{1,18}$=48.54, **P<0.001**	$F_{1,18}$=2.38, P=0.14	$F_{1,18}$=2.76, P=0.11
Pleurocarpous moss % cover	$F_{1,18}$=21.39, **P<0.001**	$F_{1,18}$=5.51, **P=0.03**	$F_{1,18}$=0.00, P=0.97
Acrocarpous moss % cover	$F_{1,18}$=0.12, P=0.73	$F_{1,18}$=0.01, P=0.93	$F_{1,18}$=2.19, P=0.16

Table 2. Mean (± SE) moss depth, index of moss microtopography, vegetation height and % cover of *Sphagnum*, pleurocarpous and acrocarpous mosses cover from ten control plots on each of two sites; and difference between cut and control plots as a percentage (± SE) of those control values. Values in cut plots were all lower than those in control plots.

Variable	Site A control	Site B control	Difference between cut and control (where cut are all < control) expressed as a percentage (±se) of control
Moss depth (cm)	11 ± 0.8	13 ± 1.5	38.0 ± 3.3
Index of moss microtopography	5.5 ± 0.2	7 ± 0.6	Site A: 12.6 ± 4.7; Site B: 32.6 ± 4.5
Vegetation height (cm)	28 ± 0.7	18 ± 0.3	62.4 ± 2.4
Sphagnum moss % cover	2 ± 0.4	37 ± 1.4	6.8 ± 10.14
Pleurocarpous moss % cover	70 ± 1.2	39 ± 1.9	13.9 ± 10.5
Acrocarpous moss % cover	5 ± 0.5	2 ± 0.3	13 ± 1.0

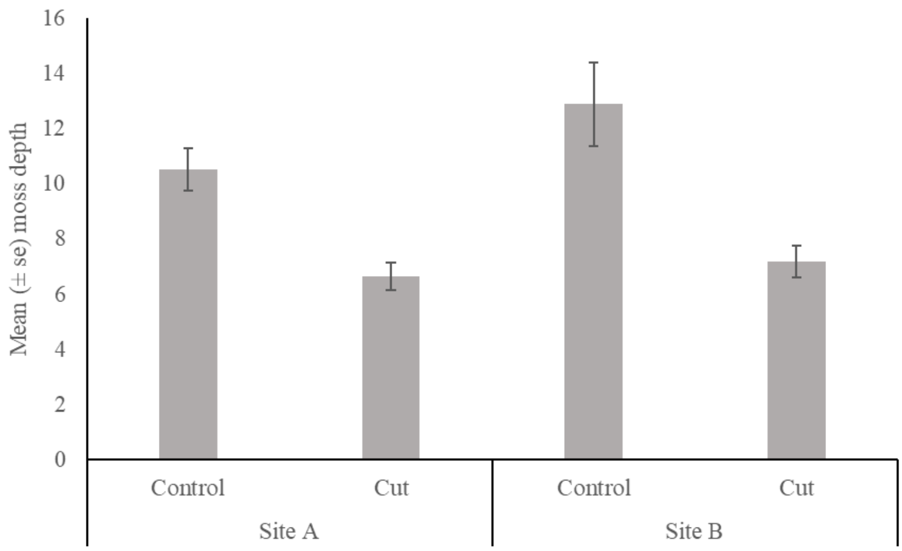

Figure 2. Mean (± SE) moss depth for cut and control plots on Site A and Site B.

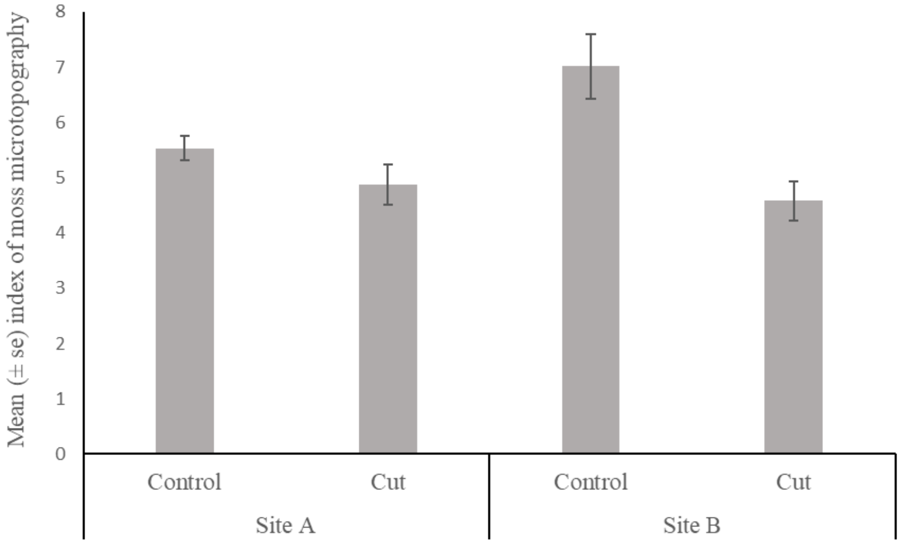

Figure 3. Mean (± SE) index of moss microtopography for cut and control plots on Site A and Site B.

Differences between sites

There was a significant effect of site on vegetation height and on percentage cover of *Sphagnum* and pleurocarpous mosses (Table 1). Comparing control plots on each site showed that vegetation height was on average 10 cm greater on Site A than on Site B. Percentage cover of *Sphagnum* mosses in control plots was nearly 20 times higher on Site B than on Site A, while pleurocarpous moss cover on Site B was half that on Site A (Table 2). We identified five species of *Sphagnum* - *S. capillifolium* (Ehrh.) Hedw., *S palustre* L., *S. medium*. Limpr., *S papillosum* Lindb. and *S. fallax* (H.Klinggr.) H.Klinggr., with the latter three found only on Site B. *S. capillifolium* was the most common, being identified in all control plots on both sites where *Sphagnum* was present. Cover of acrocarpous mosses on control plots did not vary between sites and had an average cover of just five and two percent respectively (Tables 1 and 2). Although the index of moss microtopography was higher on Site B than on Site A, (Table 2), this difference was not significant (Table 1).

DISCUSSION

The results support the hypothesis that, when considering vegetation response within the first six months after management, heather cutting resulted in an immediate reduction in moss depth and heterogeneity of moss microtopography. This effect may be due to removal of moss by cutting, or to compression by the cutting machinery. When Heinemeyer *et al*. (2019) observed some compaction of the upper peat layers after cutting, they attributed this to 'bog breathing' rather than an effect of the cutting machinery, but this assessment considered the peat layers and not the vegetation above it. In contrast, Guêné-Nanchen *et al*. (2017) observed that a reduced thickness of cultivated *Sphagnum* carpets could be partially attributed to repeat passage of mowing equipment. In the present study, although some compression of the vegetation may have occurred, visual assessment of cut plots found *Sphagnum* hummocks and areas of other mosses clearly 'scalped', leaving the remains of *Sphagnum* hummocks from which the capitula (growing tips) had been removed. Percentage cover of pleurocarpous mosses was also significantly lower on cut plots, a likely consequence of the pleurocarpous moss layer, which grows across the peat surface, being entirely removed in places. That this was not seen with *Sphagnum*, which was largely dominated by hummock-forming *S. capillifolium*, may be attributed to it forming a continuum with the peat layer, rather than sitting on the peat surface, thus retaining the basal plant structure.

Cutting also had a significant effect on moss microtopography indices on both sites, but the difference between cut and control plots was greater on Site B where control plots had higher microtopography indices than those on Site A. Despite these site differences in control plots, cut plots showed no such difference indicating that cutting reduces moss microtopography to the same level regardless of its initial status, likely to be a reflection of the standard cut-height setting of the machinery used.

Given the acknowledged importance of the moss layer, and of its hummock and hollow structure in blanket bog functioning (Aleina *et al*. 2015, Heinemeyer *et al*. 2019, Moore *et al*. 2019, Wang *et al*. 2021), these immediate impacts of cutting highlight the importance of monitoring subsequent vegetation response in the longer term. Re-establishment of the moss layer structure can be influenced by wetness, presence or absence of a mulch layer, surface topography and abundance of remnant moss (particularly *Sphagnum*) plants and fragments (Rochefort 2000). Only some moss species form hummocks (Hayward & Clymo 1982). Of these, *S. capillifolium* was the most prominent on both study sites, so factors affecting growth rate of this species are likely to be most important in determining moss structure recovery. *Sphagnum* grows from the capitulum and reproduces vegetatively through innovation of branches on the main axis into new apices (Clymo & Hayward 1982). This means that *Sphagnum*, and associated moss microtopography, may recover more quickly from any effects of compression than from cutting, because the latter can remove the growing tips. However, innovation in *Sphagnum* means it can regrow after removal of the capitulum, but the depth to which the moss is cut will determine its ability to regrow in this manner. Although innovation is possible in tissue as much as 30 cm below the surface (Clymo & Duckett 1986), Campeau & Rochefort (1996) observed highest rates in the top 10 cm, with much poorer regeneration rates from deeper material. This work also shows that mosses can regenerate from single cell fragments (diaspores) so cut material will also have the capacity to regenerate. Removal of the cut material will therefore also influence the rate of regeneration, especially when cutting removes more than 10 cm depth of *Sphagnum* mosses. Initial establishment of new moss plants may be more successful on flatter surfaces, with hummocks carrying a greater risk of desiccation (Rochefort 2000), mitigating the immediate impact of cutting on surface topographical heterogeneity. Subsequent hummock formation can be influenced by the presence of vascular plants, with ericaceous shrubs promoting moss growth both through modification of environmental conditions and physical support of the hummocks via stems and roots (Pouliot *et al.* 2011).

Evidence-based decision-making is reliant on the evidence being available. The results presented here indicate that heather cutting has an immediate impact on depth and structure of the moss layer and these short-term effects on the vegetation are likely to influence its future response. It is therefore important to continue monitoring these plots with repeat sampling over the coming years to track longer term responses. Further work is needed to determine whether altering cutting height can reduce some of the observed immediate effects of heather cutting on the moss layer, while remaining effective in its primary purpose of reducing the heather canopy. Additional research on the effects of heather cutting is also needed to consider the influence of type of machinery used, removing or leaving brash, heather age at cutting, and the effect of repeat cutting and frequency of those repeat cuts. When using burning as a heather management tool, inter-burn intervals can range from seven to 100 years (Grant *et al*. 2012, Allen *et al*. 2016, Lees *et al*. 2021), but minimum burn interval is usually dictated by the time taken for regrowing heather to reach a mature or degenerate growth stage. For cutting, there will be a trade-off between allowing plants to mature and the ability of older, more woody plants to regenerate. These research questions need to address vegetation, carbon and hydrological responses and compare them with those of burning, for which evidence gaps remain (Harper *et al*. 2018) and debate continues (Davies *et al*. 2016, Ashby & Heinemeyer 2019). To be a sustainable alternative to heather burning, the negative effects of heather cutting need to be mitigated by benefits to land management and the ecosystem, and these need to be considered in relation to the emerging evidence around the effects of burning. Only when there is adequate information on the costs and benefits of both land management practices can the best solution for the uplands, including the human communities that depend on them, be found.

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AUTHOR CONTRIBUTIONS

KH planned the work, made the field observations, wrote the first draft, and is the lead author. SW originated the work, conducted data analyses and is the corresponding author.

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