Revegetation of upland eroded bare peat using heather brash and geotextiles in the presence and absence of grazing

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SUMMARY

Revegetation of eroded bare peat is commonly facilitated by the import of artificial resources and genetic material (lime, seed and fertiliser), but such techniques are less suitable for remote upland locations with sensitive local flora. Using a BACI (Before-After-Control-Impact) approach, this study explores the effectiveness of alternative treatments (heather (*Calluna vulgaris*) brash cut onsite and two types of geotextiles) in the following four years at grazed and ungrazed sites at Ben Lawers National Nature Reserve. After an initial colonisation, the mean vegetation cover in grazed plots degraded to 9.4 %, demonstrating that restoration using these methods is impeded by trampling impacts of large herbivores. The vegetation cover and number of indicator species increased annually at the ungrazed site. A significantly greater cover (>85 %) occurred at plots where thick brash (>85 % ground cover) had been topped with GeoJute netting, but plots with only heather brash still reached 56.6 % cover. It provided a suitable seed source and colonising substrate for appropriate local peatbog species, while additional planting of *C. vulgaris* cuttings did not significantly increase vegetation cover in comparison to unplanted plots. These results show that short-term restoration of bare peat can be promoted using minimal interventions and onsite resources in the absence of grazing.

KEY WORDS: conservation management, herbivore exclusion, peatland, restoration, vegetation cover

INTRODUCTION

Peatlands are terrestrial wetland ecosystems with generally waterlogged conditions leading to an accumulation of organic matter derived from dead and decaying plant material. Their role in carbon sequestration is of global significance. Occurring in over 180 countries and covering 4.23 million km^2 worldwide (2.84 % of the land area), they account for approximately 50 % of terrestrial carbon storage, contain twice as much carbon as all forest biomass and as much as is present in the atmosphere (Gorham 1991, Parish *et al.* 2008, Xu *et al.* 2018). Peatlands therefore have a major effect on global climate regulation, as well as providing other important ecological functions including biodiversity maintenance (Erwin 2009, Littlewood *et al.* 2010), hydrological services such as regulation of water flow and water quality (Holden *et al.* 2004, Price *et al.* 2016, Shuttleworth *et al*. 2019), sediment control (Evans *et al.* 2006) and cultural benefits including aesthetic value and archaeological preservation (Bain *et al.* 2011).

While healthy, active peatlands are significant carbon sinks, their use, modification, erosion and deterioration can transform them into a carbon source (Evans & Lindsay 2010) and impede the provision of their other key ecosystem services. Peatland disturbance has occurred in many countries and has been caused by anthropogenic factors including overgrazing, burning, sulphur dioxide pollution, acidification, nitrogen deposition, drainage, afforestation, agriculture, recreation, mineral mining, energy and infrastructure development, and peat extraction (Thompson & Horsfield 1997, Tallis 1998, Vasander *et al.* 2003, Evans *et al.* 2006, Andersen *et al.* 2017, Chimner *et al.* 2017). Disturbance often alters the hydrology and vegetation composition (Stewart & Lance 1991, Austin & Cooper 2016), affecting the water quality (Anderson *et al.* 2016), lowering the soil moisture content and water table (Price 1996, Larose *et al.* 1997, Price 1997, Anderson *et al.* 2000, Holden *et al.* 2011) and leading to a loss of permanent water saturation which is essential for peat formation and carbon sequestration. In blanket peatlands in the UK, vegetation removal and gully erosion eventually result in the exposing of bare mineral ground below (Evans & Lindsay 2010). Natural processes such as water flow, frost, wind and wind-driven splash action can exacerbate degradation at a local scale (Warburton 2003, Evans & Warburton 2007). Large areas of peatlands are also predicted to be at risk from climate change effects such as water table fluctuations, moisture stress, increased soil respiration rates and wildfires (Roulet *et al.* 1992, Moore *et al.* 1998, Waddington *et al.* 1998, Zoltai *et al.* 1998, Clark *et al.* 2010). Appropriate management strategies need to be developed to enhance their resilience and ability to mitigate climate change. It has been estimated that peatland restoration in Scotland could potentially provide up to 2.7 Mt CO_2-eq savings per year by 2027 (Chapman *et al*. 2012).

A range of techniques already exist to facilitate restoration, including tree removal, gully or bare surface repair and rewetting by damming, drain blocking and ditch filling (Vasander *et al.* 1992, Quinty & Rochefort 2003, Vasander *et al.* 2003, Armstrong *et al*. 2009, Lunt *et al.* 2010, Landry & Rochefort 2012, Shepherd *et al.* 2013, Chimner *et al.* 2017, Alderson *et al*. 2019). Their aim is to improve hydrological processes, peat formation and vegetation cover, which are regarded as principal biophysical determinants of peatland function and ecosystem services (Holden *et al.* 2011). Significant investments have been made in global peatland restoration in the last 25 years, for example through the European Union's LIFE Programme and agri-environment schemes (Andersen *et al.* 2017). In northern Europe where rewetting strategies are commonly used, cut-away peatlands may return to a functional state close to that of pristine mires through large-scale collaborative projects between landowners and peat extraction companies (Vasander *et al.* 2003). Restoration work in North America has expanded to include a wide range of peatland types including bogs, fens and swamps, and has even been achieved after long-term burial by housing, golf courses, mining sediment or roads (Chimner *et al.* 2017).

Revegetation of eroded peat is beneficial for moderating flood peaks and particulate carbon loss, raising water tables, increasing biodiversity and reducing the magnitude of CO_2 losses (Waddington & Warner 2001, Grayson *et al.* 2010, Alderson *et al.* 2019). However, natural recolonisation of bare, degraded peat is impeded by its hostile microclimate of high temperatures and low levels of soil moisture, pore-water pressure and shade (Price *et al.* 1998, Sliva 1998, Price *et al.* 2003), and may progress very slowly, if at all (Salonen *et al.* 1992). Plant propagules can be introduced as soil seed banks with donor peat, vascular plant plugs transplanted from other sites or grown in nurseries, and by moss fragment transfer (Ferland & Rochefort 1997, Rochefort *et al.* 2003, Rochefort & Lode 2006, González & Rochefort 2014, Borkenhagen & Cooper 2016 Borkenhagen & Cooper 2019). Manual revegetation is also possible using application of lime, fertiliser, seeds, nurse or companion plants and mulch, which can create full plant cover in just a few years (Gore & Godrey 1981, Salonen & Laaksonen 1994, Anderson *et al.* 1997, Price *et al.* 1998, Chimner 2011, Proctor *et al*. 2013, Alderson *et al*. 2019). Such approaches involve importation of extensive artificial resources, additional nutrients and genetic material from outwith restoration sites. These can be ineffective or have harmful repercussions such as adversely affecting pH or encouraging the vigorous growth of undesirable species (Robertson 2010, Taylor *et al.* 2019). They are unsuitable for use at locations with access difficulties, minimal available funds and unique or sensitive local flora. Determining if revegetation can be promoted using onsite resources and less intensive methods will be important for encouraging and facilitating vital restoration work in such places. For example, further research is needed to assess the impact of covering peatlands with locally collected mulch or brash to stabilise eroding surfaces and encourage plant colonisation, and if the use of natural fibre geotextiles offers any additional benefits (Shepherd *et al.* 2013, Taylor *et al*. 2019).

Another knowledge gap concerns the suitability of peatland restoration techniques at sites where grazing impacts are high, as much conservation work has previously been carried out in the presence of low levels of herbivory. There was a significant increase in sheep populations in several European countries during the latter part of the 20th century (Beaufoy *et al.* 1994), including Britain, where numbers more than doubled between 1960 and 1990 (Fuller & Gough 1999). Heavy grazing by sheep (*Ovis aries*) and red deer (*Cervus elaphus*) has contributed to vegetation and community level changes in the uplands since at least the 1940s, including in peatlands and blanket bogs (Thompson *et al.* 1995, Thompson & Horsfield 1997). Completely excluding large herbivores using physical barriers such as fencing can support the restoration of grazing-sensitive montane habitats (Watts *et al.* 2019), but the effectiveness of this practice in peatlands is currently unknown (Taylor *et al.* 2019). It may also be undesirable in upland sites managed for sheep grazing or deer stalking.

This study aims to inform the conservation management and restoration of upland peatbog sites by utilising a BACI (Before-After-Control-Impact) approach at an upland site in Scotland and comparing the short-term revegetation of bare peat under the following treatments:

1. grazing (by sheep and red deer) and no grazing (within fencing);
2. for both grazing and no-grazing, application with:
   (C) heather (*Calluna vulgaris*) brash (a type of mulch) cut onsite and Coir geotextile 'netting',
   (J) heather brash and GeoJute geotextile netting, or
   (B) heather brash only;
3. application with a patchy, medium or thick cover of heather brash (for each of C, J and B); and
4. regeneration from seed contained in the brash or with the addition of planted cuttings of *C. vulgaris.*

It will analyse changes in percent vegetation cover, species richness and number of indicator species, as well as characterising which individual species are present; information that is less commonly reported in the literature (Taylor *et al.* 2019).

METHODS

The overall experimental design is as follows:

Experiment 1: a stratified randomised block design testing revegetation at thirty 2 × 2 m plots. There were 15 plots in each grazing treatment (grazing and no grazing), which comprised five separate plots of the three netting treatments (C, J and B). The heather brash cover of each plot was categorised as being one of three levels: patchy (<50 % cover), medium (50–85 %) or thick (>85 %).

Experiment 2: A complete randomised block design testing growth of planted *C. vulgaris* and revegetation at twelve 2 × 2 m plots. There were six within each grazing treatment (grazing and no grazing), comprising two separate plots of each of the three netting treatments (C, J and B). All plots contained four planted *C. vulgaris* cuttings and a medium cover (50–85 %) of brash.

Study site

The study was conducted at the Ben Lawers Natural Nature Reserve (NNR), which is situated in the Southern Highlands of Scotland (56° 30′ 39″ N, 4° 15′ 45″ W) and owned and managed by the National Trust for Scotland (NTS). The underlying geology consists of calcareous Dalradian mica schist. It is arguably the most important site in Britain for arctic-alpine flora (Mardon & Watts 2019) and has also been designated as a Site of Special Scientific Interest (SSSI), a National Scenic Area (NSA) and a Special Area of Conservation (SAC). The primary habitats present at Ben Lawers and listed in Annex 1 of the European Union Habitat Directives (European Commission 2013) include alpine calcareous grasslands, alpine heath, the hydrophilous tall herb fringe community and blanket bog. The latter is distributed across the site but has previously been allocated an unfavourable conservation status due to the effects of erosion and grazing by large herbivores (Watts 2013). As a result, two locations were chosen for experimental peat restoration: (1) Coire Odhar (CO), which was grazed predominantly by sheep from nearby farms with heritable grazing rights (at least 0.4 per ha) and also open to access by red deer at a much lower density of 8.2 per km^2 in 2015 (BDMG 2016); and (2) the southern ridge of Meall nan Tarmachan (MT), where sheep have been excluded by fencing since 2007 and red deer since 2011. Prior to restoration, both sites contained a large number of patches of eroded bare peat. They were approximately 3 km apart and situated on a south-facing or south-east facing aspect within an altitudinal range of 664–773 m above sea level, and so were very similar to each other except for grazing management.

Experiment 1: heather brash and geotextile netting

In autumn 2014 a Softtrak all-terrain rubber tracked vehicle fitted with a forage harvester was used to cut a supply of heather brash from a well vegetated, characteristically peaty area within the Tarmachan fence. This time of year was chosen to ensure the inclusion of a good seed source of heather and other bog plants. The brash was then airlifted and dropped next to areas of eroding peat at the MT and CO sites. By early spring 2015 it had been transferred into flexible plastic trugs and spread by hand over the bare peat, with a single 1 m^3 bag of brash covering an area of approximately 50 m^2. Once on the ground, the heather brash covered a depth of no more than 1–2 cm and allowed little light penetration when applied thickly. It was composed of fine material such as leaves, slender branches and the previous season's flower shoots, as well as occasional thicker stems.

After layering with brash, individual patches of bare peat were then randomly allocated one of three additional treatments: covering with 400 g m^{-2} biodegradable Coir netting (C), covering with 500 g m^{-2} biodegradable GeoJute netting (J) or no netting (i.e., brash only (B)). The bales of netting were airlifted or transported uphill by an amphibious off-road vehicle, then rolled out on top of the brash and secured using approximately three steel pins per m^2. Small islands of un-eroded peat hummocks were sometimes removed to make laying of geotextiles easier and flatter. This removed material was utilised in the blocking of drainage channels in other parts of the restoration sites.

Across both CO and MT, five 2 × 2 m study plots were set up in Spring 2015 using stratified random placement on separate patches of 100 % bare peat with each of the three treatment types (C, J or B; 30 experimental units in total). All four corners were positioned with 15 cm nails pushed into the peat and left in situ. There was some variation in the thickness of the heather brash spreading, which was noticed when the plots were being permanently marked. To account for this unevenness, an extra category was introduced into the treatments, with each plot classified into one of three levels of brash cover: patchy (<50 % cover), medium (50–85 %) and thick (>85 %). At CO there were five plots with each level of brash cover, but at MT there were four patchy, five medium and six thick in total (Table 1).

Five untreated 2 × 2 m patches of bare peat were also monitored at each restoration site as a control. All plots were surveyed annually in July 2016–2019 to assess revegetation for the four years following treatment. A metal detector, Global Positioning System navigation device (with Glonass) and site photographs were used to aid plot re-location to the exact same position. The percent cover of all species present in the plots was recorded using a visual estimation. Errors were minimised by using the same observer throughout the study and keeping to a maximum of five plots per day to reduce observer fatigue and drift during surveying. The presence of hoofmarks within the plots or damage to the netting was also noted.

Experiment 2: planted heather

Cuttings of heather (*C. vulgaris*) were taken from the MT brash harvesting site in autumn 2014 and pricked out into small plug trays for growing on within a local tree nursery. Across the three peat treatment types (C, J or B), 86 of these cuttings were planted at CO in summer 2016 and 111 at MT, avoiding patches already used for the 30 study plots in Experiment 1. For Experiment 2, a complete randomised block design was set up using twelve 2 × 2 m plots, each containing four *C. vulgaris* cuttings. This planting density is roughly similar to that used in montane woodland and scrub restoration which has been ongoing elsewhere at Ben Lawers since 1987. At both sites (CO and MT) there were two replicates on each of the three netting treatments (C, J and B). Brash cover was medium (50–85 %) at all plots and they were permanently marked with 15 cm nails. The percent cover of vegetation was recorded using a visual estimation, and, for each *C. vulgaris* cutting planted in the plots, the height, longest canopy diameter and the diameter horizontally perpendicular to it were determined using a metal ruler. These measurements were repeated in July 2018 and 2019, when the survival of all *C. vulgaris* cuttings planted across the two restoration sites was also recorded.

Table 1. Numbers of plots at each restoration site, Coire Odhar (CO) and Meall nan Tarmachan (MT), with the different netting treatments (C = Coir and heather brash; J = GeoJute and heather brash; B = brash only) and levels of brash cover defined as patchy (<50 % cover), medium (50–85 %) and thick (>85 %).

	CO				MT		
Treatment	C	J	B		C	J	B
Patchy	3	1	1		1	2	1
Medium	1	2	2		1	2	2
Thick	1	2	2		3	1	2

Data analysis

The percent cover of all species present in each plot was combined to give an estimation of total revegetation cover. The total species richness (number of species present in each plot) was also calculated, combining closely related species at the generic level that were difficult to separate consistently without causing disturbance to the sensitive vegetation (*Agrostis* spp. and *Sphagnum* spp.) The total number of blanket bog indicator species defined by the UK Joint Nature Conservation Committee (JNCC 2009) and listed in Appendix 1 was also calculated for each plot. Nomenclature follows Stace (2019) for vascular plants and Atherton *et al*. (2010) for bryophytes.

In the absence of any revegetation at untreated patches of bare peat (control plots), all analyses apply to treated plots only. For Experiment 1, a repeated measures mixed effects model (ANOVA) was used to test for the effects of site, netting type (C, J or B), brash cover and year on the total percent vegetation cover, the number of indicator species, total species richness and the percent cover of functional species groups (bryophytes, graminoids and shrubs). The cover of herbs and lichens was too low to be tested in this way. Variation between plots was included as a random effect and nested within the site factor. The restricted maximum likelihood (REML) approach was used for variance estimation and the Kenward-Roger approximation for denominator degrees of freedom.

For Experiment 2, the proportional percentage change in the three measurements of each surviving planted *C. vulgaris* in the twelve study plots was calculated using the formula $((x - y) / y) \times 100$ where x is the 2019 measurement and y is the 2016 measurement. Variation between the two sites was tested using *t*-tests, because General Linear Models (GLMs) run with netting as a nested factor found that the effect of this treatment was not significant. By comparing data from the 30 plots in Experiment 1 (unplanted) and the 12 plots in Experiment 2 (planted with *C. vulgaris*), a GLM was used to test for the effects of site, netting type (C, J or B) and planting treatment (unplanted or planted) on the total percent vegetation cover in 2019 (four years after treatment).

Non-significant interactions were removed from the models using the Akaike Information Criterion. Significant differences between treatments were identified using post hoc Tukey Pairwise Comparisons. The normality of the original data and the residuals was assessed using the Kolmogorov–Smirnov test and histograms, and the assumption of equal variance was determined using Levene's test.

RESULTS

Experiment 1: percent vegetation cover

The observations at untreated patches of bare peat found that no spontaneous revegetation had occurred in the absence of any restoration work at either study site. The mixed effects model (ANOVA) of the total percent vegetation cover showed that site had a significant effect on the response to treatment of bare peat, and that there was also a highly significant interaction between the site and year factors (Table 2). In the first year after treatment there was a similar amount of revegetation at the two sites (mean = 27.9 %, SE = 5.18, n = 30) but over the next three years they followed contrasting trajectories (Figure 1). At MT, the ungrazed site, there was a significant increase towards a mean cover of 52.4 % (SE = 8.58), whereas at CO, the grazed site, there was a significant decrease towards a mean cover of 9.4 % (SE = 4.91).

Brash cover also had a significant effect (Table 2), with thick, medium and patchy levels of application producing greater, intermediate and lower vegetation cover respectively. In addition, there were significant interactions between the brash, year and site factors, and the netting type, year and site factors (Table 2). At MT, the total vegetation cover in plots with patchy brash or Coir netting (C) did not change during the four years following treatment, but there was an increase with time where thick and medium brash were applied (Figure 2, (a) and (c)). Vegetation cover also increased over all four years on GeoJute (J), but

Table 2. The results of repeated measures mixed effects models (ANOVAs), showing *P*-values for percent vegetation, bryophyte, graminoid and shrub cover, number of indicator species and total number of species. Factors significant at the 95% confidence level are highlighted in bold. Non-significant interactions have been removed using Akaike Information Criterion to give the best fitting model to the data.

Factor	Df	Total % vegetation cover	No. indicator species	Total species richness	% bryophyte cover	% graminoid cover	% shrub cover
Site	1	**0.005**	**<0.001**	**0.004**	**0.019**	**0.037**	0.275
Netting type	2	0.595	0.825	0.363	0.478	0.562	0.842
Brash cover	2	**0.046**	0.483	0.107	0.076	0.182	0.841
Year	3	0.450	**0.023**	0.134	0.230	**<0.001**	**<0.001**
Site*Netting type	2	0.438	0.922	0.192	0.658	0.075	0.540
Site*Brash cover	2	0.145	0.829		0.218	0.725	0.275
Site*Year	3	**<0.001**	**0.018**	**<0.001**	**<0.001**	**<0.001**	0.171
Netting type*Brash cover	4	0.315	0.998		0.282	0.674	0.835
Netting type*Year	6	**0.019**	0.570		**0.046**	0.931	0.981
Brash cover*Year	6	**0.018**	0.747		**0.004**	0.909	0.852
Site*Netting type*Year	6	**0.006**	0.301		**0.059**	0.061	0.422
Site*Brash cover*Year	6	**0.036**	0.306		0.364		0.094
Netting type*Brash cover*Year	12	**0.022**			**0.024**	0.576	
Site*Netting type*Brash cover	4				0.993	0.321	
Site*Netting type*Brash*year	12				0.287		

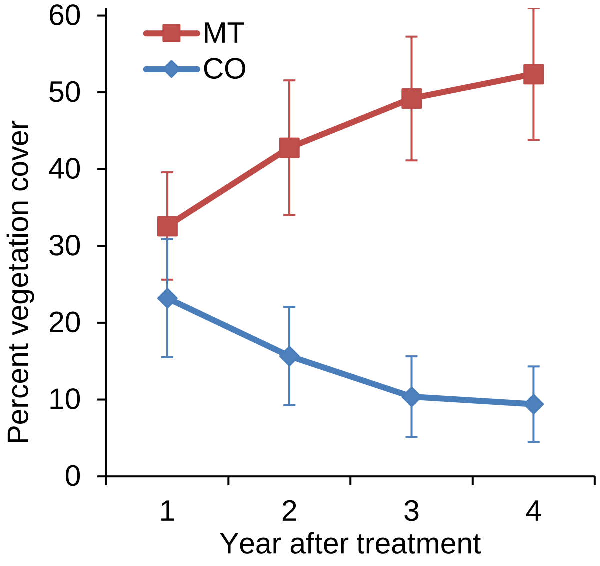

Figure 1. The mean percent vegetation cover in the four years following treatment at the plots at Meall nan Tarmachan (MT) n = 15 and Coire Odhar (CO) n = 15. Error bars are ± 1 standard error (SE).

only between years 1 and 2 at the brash-only plots (B). Conversely, the revegetation in plots at CO with thick brash or J netting was initially higher in the first year but then declined over time to correspond with the cover in plots with medium and patchy brash or C and B treatments, which did not show any change over time (Figure 2, (b) and (d)).

Thus, the greatest levels of revegetation (at least 85 %) were recorded after four years on thick brash covered with GeoJute (J) at the site MT associated with the absence of grazing. One plot treated in this way reached 100 % vegetation cover and another 99 %. In contrast, all netting and brash treatment types in the presence of grazing had similarly low vegetation cover after four years. However, there was some variation in the revegetation of plots within treatment types, the most notable example being a C plot at CO with patchy brash which had approximately 80 % cover during each of the four years following treatment.

Experiment 1: number of species

Site had a significant effect on the number of indicator species and total species richness in the treated plots, and interacted with the year factor (Table 2). The number of indicator species remained constant with time at CO but increased over the four years following treatment at MT (Figure 3). In contrast, total species richness was similar at both sites in years 1 and 2, but thereafter increased at MT and decreased at CO.

Experiment 1: species groups

The dominant vegetation group across all plots and all years was bryophytes (Figure 4), primarily *Campylopus flexuosus*, *Pleurozium schreberi*, *Polytrichum strictum*, *Hylocomium splendens*, *Rhytidiadelphus loreus* and *Sphagnum* spp. (Table 3). A list of all species recorded in the plots throughout the study is given in Appendix 2. Percent bryophyte and graminoid cover were greater at MT and showed an interaction between the site and year factors (Table 2). At MT, graminoid cover increased in years three and four after treatment but bryophyte cover remained similar over time (Figure 4), apart from an increase at brash only (B) sites between years 1 and 2. Conversely, bryophyte cover declined over time at CO (particularly on J netting and where thick brash was applied) and graminoid cover showed no change (Figure 4). Additionally, the dominant graminoid species at MT were *Eriophorum vaginatum* and *Avenella flexuosa*, whereas *Carex echinata*, *Agrostis* spp. and *Festuca* spp. were more highly represented at CO (Table 3).

Shrub cover only varied between years (Table 2), increasing three and four years after treatment (Figure 4). It was almost exclusively composed of *Calluna vulgaris*, with the addition of very small amounts of *Empetrum nigrum* and *Vaccinium myrtillus* in a few MT plots (Table 3). Throughout all four years the cover of herbs and lichens was very low, particularly at CO (Figure 4). However, lichen cover and diversity at MT increased after three and four years; mostly accounting for the rise in total species richness shown in Figure 3. These were primarily *Cladonia* spp., including *Cladonia chlorophaea* agg., *Cladonia uncialis*, *Cladonia portentosa*, *Cladonia furcata* and *Cladonia bellidiflora* (Table 3). Thus, at MT the increase in overall percent vegetation cover with time corresponded with a rise in the cover of graminoids, shrubs and lichens (but not bryophytes).

Experiment 1: disturbance and site damage

All CO plots were subjected to significant trampling by hooves, even in the first year following treatment (Table 4). The two types of netting responded differently to this disturbance. The GeoJute (J) remained adhered to the peat surface but became increasingly unravelled, while the Coir (C) stayed intact but was unpinned and pulled off by the trampling action of grazing animals to reveal the bare peat below. Only one Coir plot at CO still had less than 50 % netting pulled off in the 4th year, and this plot also had a much greater vegetation cover (75 %) than any other plot at that site. In contrast, the netting at all MT plots remained pinned onto the peat after

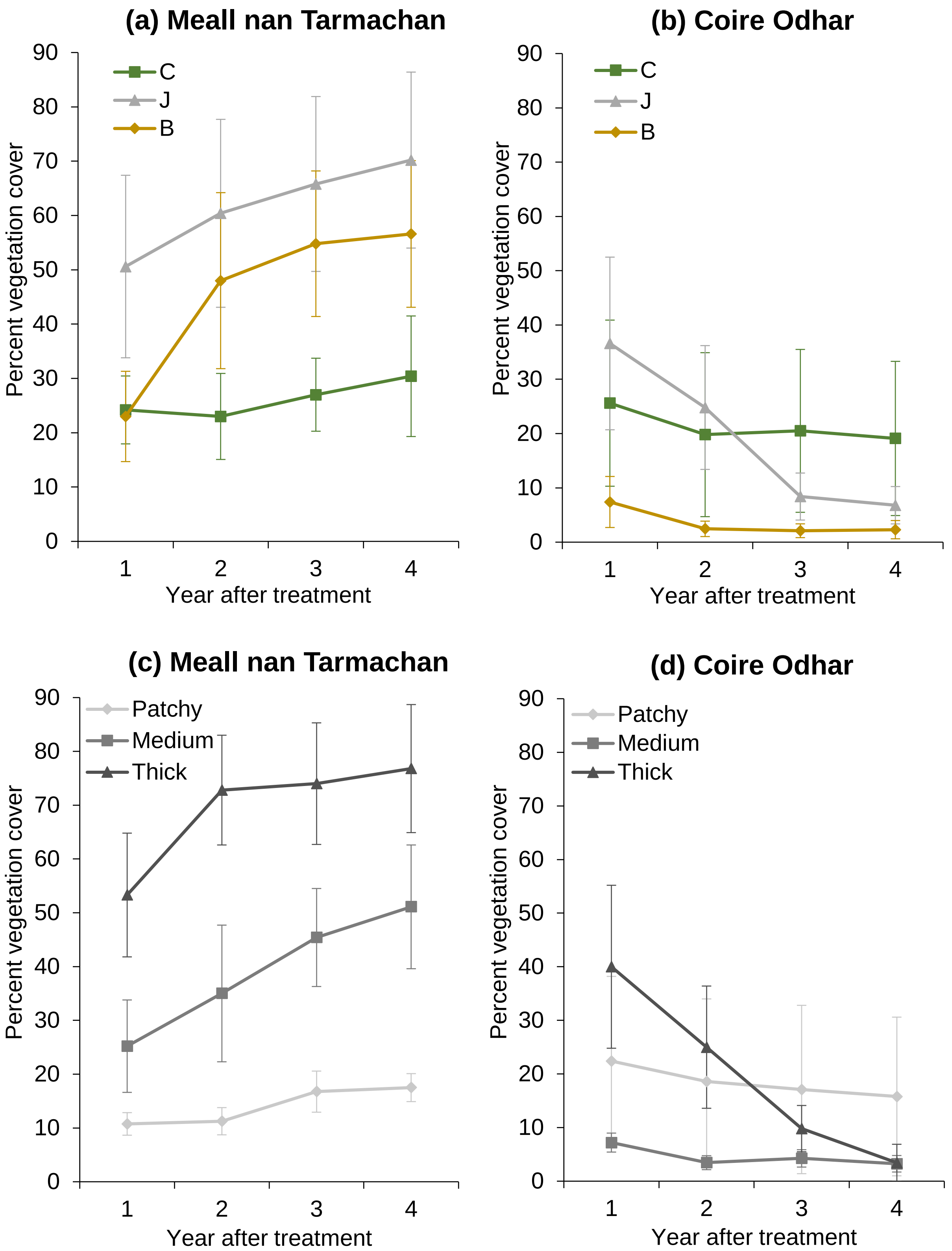

Figure 2. The mean percent vegetation cover in the four years following treatment for the plots at (a) Meall nan Tarmachan (MT) applied with three different netting treatments, (b) Coire Odhar (CO) applied with three different netting treatments, (c) Meall nan Tarmachan (MT) applied with three levels of brash cover* and (d) Coire Odhar (CO) applied with three levels of brash cover*. Error bars are ± 1 SE. C = Coir and heather brash; J = GeoJute and heather brash; B = brash only. *brash cover defined as patchy (<50 % cover), medium (50–85 %) and thick (>85 %).

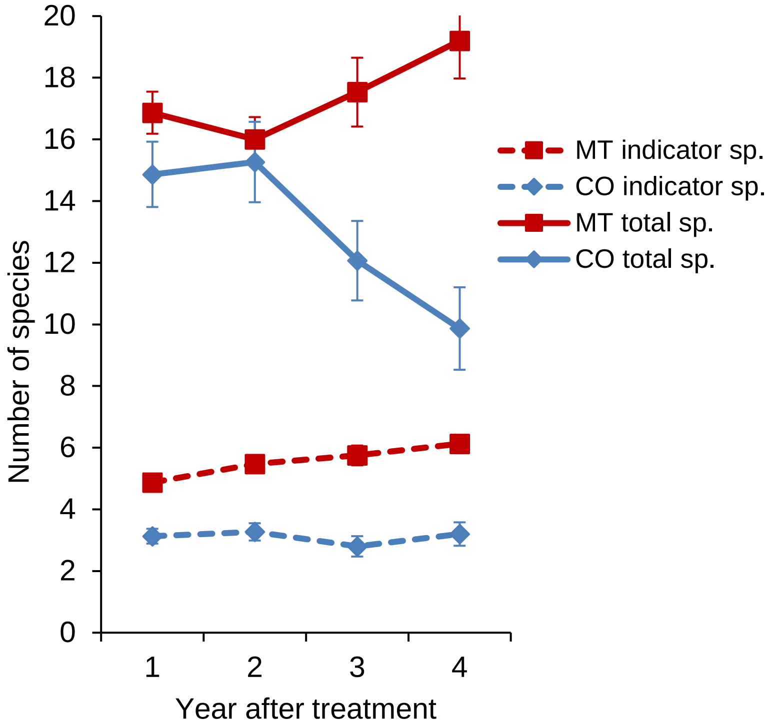

Figure 3. The mean total number of species (richness) and mean number of blanket bog indicator species recorded in the plots at Meall nan Tarmachan (MT) and Coire Odhar (CO) in the four years following treatment of bare peat. Error bars are ± 1 SE.

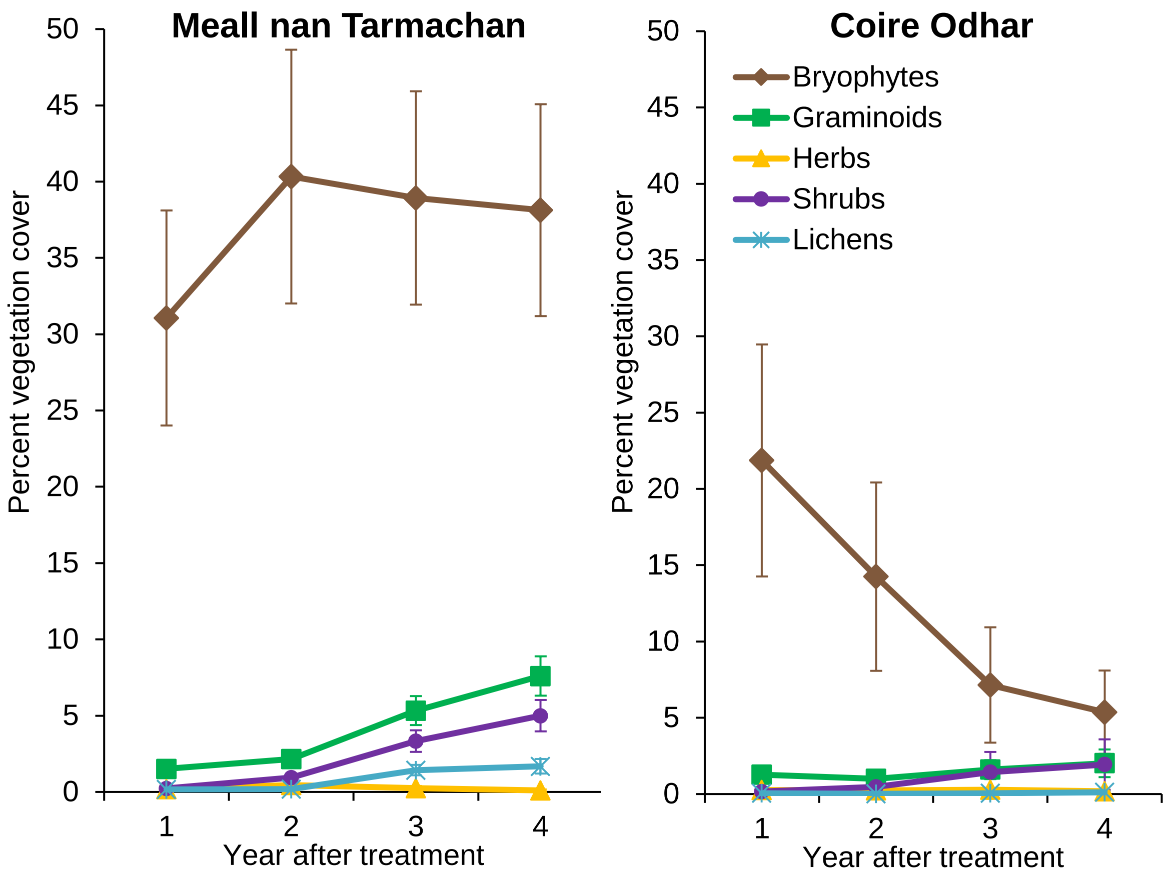

Figure 4. The mean percent cover of different species groups recorded in the plots at Meall nan Tarmachan (MT) and Coire Odhar (CO) in the four years following treatment of bare peat. Error bars are ± 1 SE.

Table 3. Frequency (number of occupied plots) and mean percent cover of species at CO (Coire Odhar) and MT (Meall nan Tarmachan) in the fourth year after treatment (2019). Only species that were recorded in at least three plots at either site are given, and mean percent cover has been calculated for those plots in which they occurred. Species are arranged in descending order of frequency and grouped by species group (GR: graminoids, HB: herbs, SH: shrubs, BR: bryophytes, LI: lichens).

Species name	Species group	Total plots	No. CO plots	CO plot mean	No. MT plots	MT plot mean
Eriophorum vaginatum	GR	14	1	1.00	13	4.96
Avenella flexuosa	GR	10	0	0.00	10	2.52
Festuca ovina/vivipara	GR	9	4	0.63	5	0.75
Eriophorum angustifolium	GR	7	4	0.63	3	3.00
Agrostis spp.	GR	6	5	0.75	1	0.25
Carex echinata	GR	5	4	1.63	1	0.25
Luzula multiflora	GR	4	3	0.25	1	0.25
Galium saxatile	HB	8	5	0.55	3	0.50
Potentilla erecta	HB	3	3	0.25	0	0
Calluna vulgaris	SH	20	7	4.14	13	5.36
Vaccinium myrtillus	SH	3	0	0.00	3	0.25
Hylocomium splendens	BR	29	15	0.72	14	3.95
Polytrichum strictum	BR	29	15	1.57	14	3.95
Pleurozium schreberi	BR	26	12	0.96	14	6.68
Sphagnum spp.	BR	24	10	1.98	14	2.45
Campylopus flexuosus	BR	23	9	0.83	14	16.60
Rhytidiadelphus loreus	BR	23	10	0.68	13	1.25
Aulacomnium palustre	BR	23	10	0.40	13	0.88
Plagiothecium undulatum	BR	19	8	0.34	11	1.04
Dicranum scoparium	BR	18	9	0.25	9	0.58
Racomitrium lanuginosum	BR	13	3	0.25	10	0.86
Polytrichum commune	BR	8	2	0.63	6	0.67
Lophozia ventricosa	BR	7	0	0.00	7	0.86
Rhytidiadelphus squarrosus	BR	4	1	0.25	3	0.75
Cladonia portentosa	LI	14	3	0.25	11	0.38
Cladonia chlorophaea agg.	LI	13	0	0.00	13	0.52
Cladonia uncialis	LI	12	0	0.00	12	0.42
Cladonia bellidiflora	LI	11	0	0.00	11	0.32
Cladonia furcata	LI	10	0	0.00	10	0.45
Cladonia arbuscula	LI	6	0	0.00	6	0.25

Table 4. Observations of hoof impact and netting damage across the 15 plots at each restoration site, Coire Odhar (CO) and Meall nan Tarmachan (MT), in the four years following treatment with heather brash and geotextiles. C = Coir netting; J = GeoJute netting.

Damage type	CO					MT			
	1	2	3	4		1	2	3	4
>25 hoof marks	12	15	15	15		0	0	0	0
>50 % netting unravelling	2 J	4 J	5 J	5 J		0	0	1 J	1 J
>10 % netting pulled off	3 C	3 C	2 C	1 C		0	0	0	0
>50 % netting pulled off	2 C	2 C	3 C	4 C		0	0	0	0

four years, with the GeoJute becoming noticeably unravelled at only one plot (Table 4). Figure 5 provides photographic examples of the vegetation cover and damage at a contrasting location from each site.

Experiment 2: heather planting

Three years after planting, the survival rate of the *C. vulgaris* cuttings was 31.4 % at CO and 84.7 % at MT. Significantly more had therefore survived at the ungrazed site than at the grazed site (χ^2 = 14.90, P < 0.001). There was also a significant difference between the two sites for mean percentage change in height (t = 5.94, P <0.001), length (t = 6.71, P < 0.001) and width (t = 6.26, P < 0.001) of the surviving plants. They had increased in size at MT but decreased at CO (Figure 6). However, there was no difference in vegetation cover at the planted and unplanted plots (Figure 7, Table 5).

DISCUSSION

Experiment 1: percent vegetation cover and number of species

This study found significantly greater revegetation of bare peat four years after treatment at the ungrazed site (MT; mean plot cover 52.4 %) than the grazed site (CO; mean plot cover 9.4 %). All plots open to sheep and red deer sustained significant impact by hooves, which progressively degraded the fragile colonising vegetation, particularly where only heather brash was applied. This is because plants growing on waterlogged peaty soil are easily damaged by trampling, even at low grazing intensities (Pellerin *et al.* 2006). The netting offered no additional protection from large herbivores after the first year, although both types responded very differently. The GeoJute (J) became increasingly unravelled, while the Coir (C) was more durable but prone to dislodging to reveal bare peat. Where it remained pinned to the ground surface the Coir facilitated the colonisation of some localised patches of revegetation (up to 80 % cover). Nevertheless, these results indicate that restoration of upland peatlands with sheep densities of 0.4 per ha or more is challenging and cannot be achieved using the methods described in this study. The work undertaken at Ben Lawers NNR will not be applicable to degraded peatbog sites with even greater densities of sheep. Lighter grazing (0.1–0.37 sheep per ha) could be a more appropriate management strategy, but total exclusion may be required for the first 3–5 years of restoration projects and up to 10 where damage is severe (Rawes & Hobbs 1979, Lunt *et al.* 2010). Blanket bog systems are unlikely to sustain heavy grazing for long without considerable degradation occurring (Rawes & Hobbs 1979). Reporting such "failures" (interventions that have no meaningful effect) is important for minimising the use of ineffective peatland restoration techniques elsewhere (Taylor *et al.* 2019).

The amount of heather brash applied to bare peat also had a significant effect on restoration outcomes, with thick coverage (>85 %) producing greater revegetation than medium levels (50–85 %), and patchy brash (<50 %) the least. This result agrees with Rochefort *et al.* (2003), who found that a continuous layer of peatland plant material 1–2 cm thick allowed better plant establishment than a scant layer after four growing seasons, and that plots protected with straw mulch had a greater percent cover than those without. This outcome is expected because brash and mulch offer a source and substrate for colonising plants (Buckler *et al.* 2013). Such plant litter also has the potential to improve soil conditions by regulating and stabilising temperatures, humidity, water loss and light levels (Carson & Peterson 1990, Facelli & Pickett 1991, Xiong & Nilsson 1999, Groeneveld *et al.* 2007, Chimner 2011). The presence of shade is important for the regeneration success of peatland bryophytes (Graf & Rochefort 2010). Peat

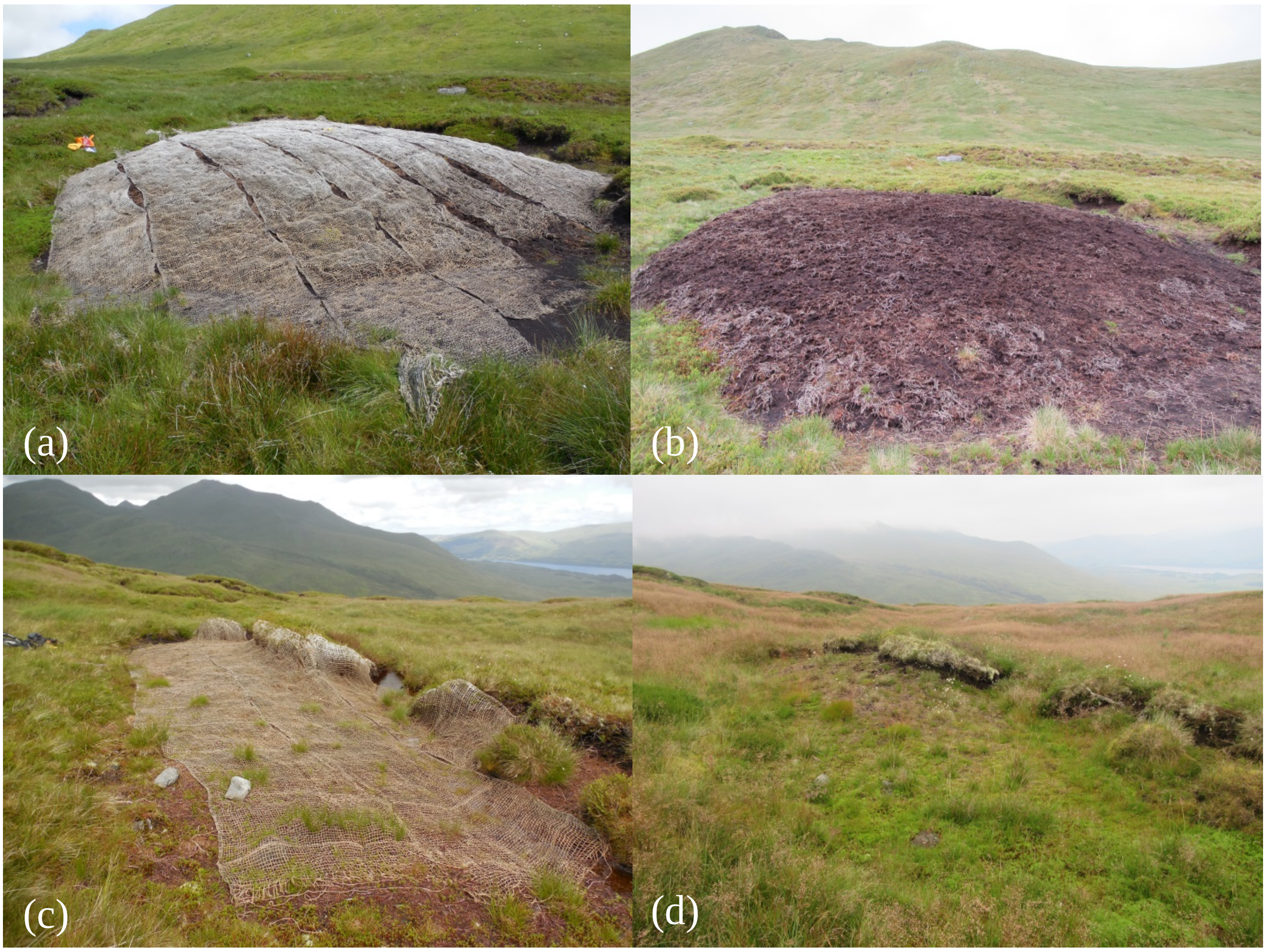

Figure 5. A photographic comparison of the revegetation and netting damage sustained by two sites applied with thick heather brash and Geojute netting. (a) Coire Odhar (grazed) site just after treatment, (b) the same CO site four years later; (c) Meall nan Tarmachan (ungrazed) site one year after treatment (d) the same MT site three years later.

restoration projects utilising heather brash should therefore concentrate on spreading it to at least an 85 % ground coverage, rather than more thinly over a larger area. However, applications with depths of more than 1–2 cm are probably unnecessary and may promote a community composed mainly of vascular plant species (Rochefort *et al.* 2003, Buckler *et al.* 2013).

GeoJute netting also significantly improved revegetation after four years in the absence of grazing by restoring a mean cover of 70 %. This outcome is similar to that found by the Royal Society for the Protection of Birds (RSPB) at the Hobbister nature reserve on Orkney (a lowland, coastal site in the north of Scotland), where peat addition plus mulch and GeoJute was the most effective treatment after three years and produced 80 % vascular plant cover (Robertson 2010). Netting is useful because it acts as a soil stabiliser and provides brash and bare peat with a protective microclimate to buffer the effects of erosion, wind ablation, moisture loss and surface runoff (Buckler *et al.* 2013). Although less revegetation occurred on the Coir netting plots at MT, this may be because nearly all had only medium or patchy brash; further demonstrating that the amount of brash has a stronger influence on restoration outcomes than netting type.

Although spontaneous revegetation of bare peat has been recorded in other situations and is relatively common in erosion gullies (Lavoie *et al.* 2003, Evans & Warburton 2007, Harris & Baird 2019), it was not observed at either restoration site at Ben Lawers NNR. Timescales far longer than four years may be required in exposed locations where natural processes, in particular wind, are maintaining bare surfaces. This highlights the need for human intervention to restore blanket bog vegetation and function in upland environments.

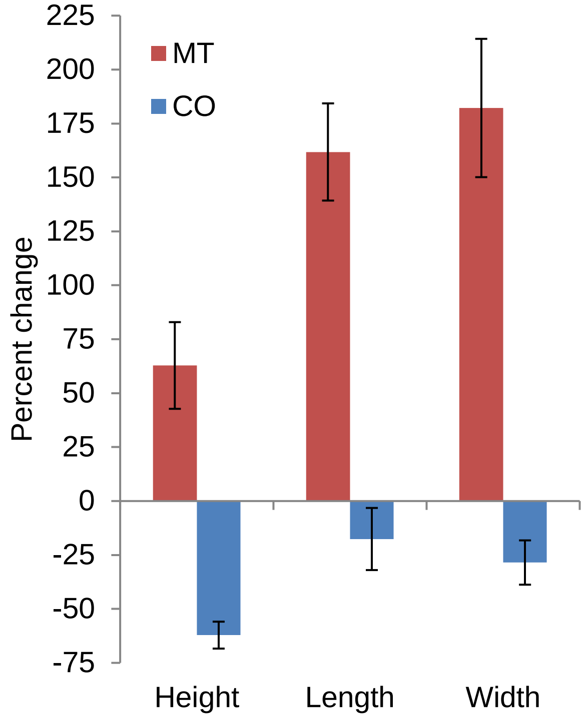

Figure 6. The mean percentage change in height, length and width of surviving *Calluna vulgaris* cuttings three years after planting. $n = 24$ for Meall nan Tarmachan (MT) and $n = 8$ for Coire Odhar (CO). Error bars are ± 1 SE.

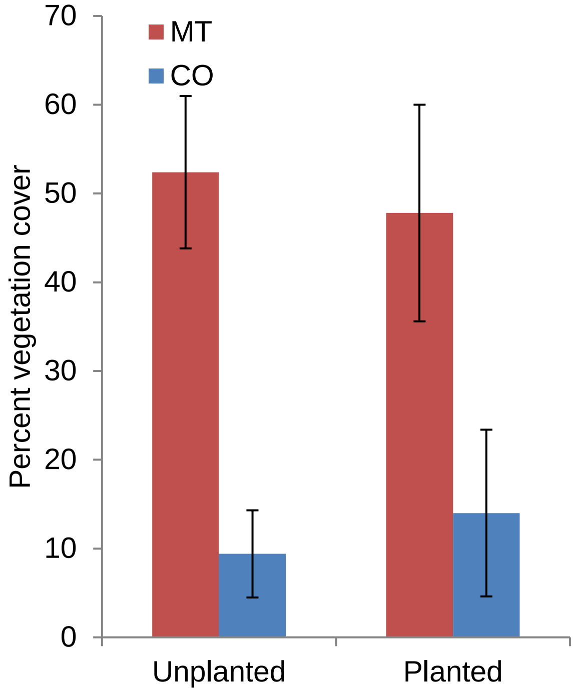

Figure 7. The mean percent vegetation cover in 2019 (four years after treatment with heather brash) in the unplanted plots ($n = 30$) and plots planted with *Calluna vulgaris* ($n = 12$) in 2016 at Meall nan Tarmachan (MT) and Coire Odhar (CO). Error bars are ± 1 SE.

Peatbog restoration projects in Britain sometimes employ a LSFM (lime, seed, fertilizer, mulch) approach; application of lime, fertiliser, sown seed and nurse plant plugs in addition to mulch and geotextile cover (Lunt *et al*. 2010). Alternative techniques developed elsewhere, for example in North America, include transfer of propagules collected from undisturbed donor bogs, protection with straw mulch and phosphorus fertilisation (Rochefort *et al*. 2003, Sottocornola *et al*. 2007, Graf & Rochefort 2008). Use of intensive LSFM methods is common practice in the English Peak District, and can convert bare peat to a full vegetation cover after four years (Proctor *et al*. 2013). Alderson *et al*. (2019) also showed that LSFM in the English Pennines produced nearly 75 % vegetation cover after two years and indicator species plateaued at approximately four per 2 × 2 m plot after 12 years. This work provided evidence of a shift towards a stable moorland community. With only geotextiles and brash, the vegetation cover at MT exceeded 90 % at several plots, and the mean number of indicator species was six after just four years. Brash alone (a much cheaper and less labour-intensive option) still produced a mean vegetation cover of 57 % at MT (maximum 97 %). This study therefore demonstrates that revegetation of bare peat with species appropriate to the blanket bog habitat can be effectively achieved in the absence of grazing with minimal intervention if heather brash can be

Table 5. GLM testing of percent vegetation cover in the plots at Coire Odhar (CO) and Meall nan Tarmachan (MT) in 2019; four years after treatment with heather brash and different netting types (C, J or B). Planting treatments are unplanted ($n = 30$) or planted ($n = 12$) with *Calluna vulgaris* cuttings in 2016. Interactions removed from the full model during simplification process: Site*Planting, $F = 0.31$, $P = 0.583$; Netting*Planting, $F = 0.21$, $P = 0.814$; Site*Netting*Planting, $F = 0.67$, $P = 0.521$.

Factor	Df	MS	*F*	*P*
Site	1	17141.7	32.6	**<0.001**
Netting type	2	1616.4	1.54	0.229
Planting	1	<0.1	<0.01	0.994
Site*Netting	2	3898.9	7.42	**0.002**
Error	35	18403.1		

harvested onsite and applied with sufficient ground coverage. Such an approach will be particularly important in upland locations such as Ben Lawers NNR where conservation work is time consuming and challenging due to remoteness, access difficulties and the sensitivity of the montane environment.

Experiment 1: species groups

Bryophytes were the dominant plant group restored to the treated peat, undoubtedly because of their totipotency; the ability to regenerate a whole plant from a single cell (Lal 1984). They are usually the first species to naturally colonise degraded or damaged ground because only fragments are required for establishment (Proctor *et al.* 2013), in contrast to most vascular plants which need seed and time for their germination. The species composition of colonising bryophytes corresponds with what is expected for the habitat. For example, *Polytrichum strictum* may aid restoration by creating a more favourable microenvironment for bog plant growth, thereby acting as a nurse plant and facilitating the return of the *Sphagnum* layer (Groeneveld & Rochefort 2002, Groeneveld *et al.* 2007). These are the mosses primarily responsible for peat formation and are crucial for restoration success in terms of carbon sequestration (Lunt *et al.* 2010) and reducing overland water flow velocities, flooding and erosion (Holden *et al.* 2008). Spontaneous regeneration of *Sphagnum* on bare peat is exceptional and requires long timescales (Price & Whitehead 2001, Chapman *et al.* 2003), but this study has demonstrated that colonisation can be encouraged using heather brash alone or with netting, particularly in the absence of grazing. *Sphagnum* mosses are very sensitive to damage and unlikely to withstand more than one or two trampling events in a year (Lindsay *et al.* 2014a). However, even at the ungrazed site the mean cover of *Sphagnum* after four years was generally low (2.5 %). A moss fragment transfer approach specifically targeting these species or micropropagation in the form of beads, plugs or gel offer the potential for significantly greater establishment of *Sphagnum* over short timescales (Ferland & Rochefot 1997, Rochefort *et al.* 2003, Caporn *et al.* 2018). The introduction of alien material can be avoided by generating these propagules from local sources, but such methods may be more intensive and costly than those used here.

Although bryophytes were the primary colonisers, they showed no further increase after the first year. An expansion of graminoid, shrub and lichen cover accounted for the increase in overall percent vegetation cover at MT with time. *Calluna vulgaris*, a prolific seed producer, will have sprouted from seed present in the brash, with individuals becoming more robust each year, including those at CO which produced a similar cover to MT. They may have avoided direct grazing impacts by being small and hidden amongst the bryophyte ground layer, unlike the larger planted plugs. However, the species composition of graminoids was different at the two sites, suggesting that they colonised from seed dispersing from the surrounding area, rather than in the brash. This difference is because the vegetation growing at each location relates to the presence or absence of large herbivores which significantly affects light availability (Watts *et al.* 2019). *Avenella flexuosa* is more abundant inside the large herbivore exclosure as it produces moderately large seeds that may establish in closed vegetation and seedlings that can persist and thrive even in shady conditions (Grime & Jeffrey 1965, Hill *et al.* 1992, Watts *et al.* 2019). *Eriophorum vaginatum* (frequent at MT) performs less well in response to grazing pressures than *Festuca ovina/vivipara* or *Agrostis* spp. (Pollock *et al.* 2007), which were more common at CO and can decrease in abundance if herbivores are removed (Watts *et al.* 2019). The resilience of these species under heavy grazing explains why graminoid cover, although low, did not decline over time at CO, thus demonstrating the value of using local species during bare peat revegetation.

At MT there was an increase in lichen cover and diversity (*Cladonia* spp.) over time, which mostly accounted for the rise in the total number of species. These lichens are particularly sensitive to grazing by livestock and disturbance by deer (Rawes & Hobbs 1979; Fryday 2001, Pellerin *et al.* 2006), and the growth rates of lichens in bogs are even slower than those of *Sphagnum* mosses (Vasander 1981). An increase in terricolous lichen cover and diversity is generally associated with enhanced soil health and structure, and they are therefore key indicators of ecosystem stability and function (Will-Wolf *et al.* 2002). This further highlights the improved restoration outcomes at MT, in contrast to CO.

Experiment 2: heather planting

The survival rate of the planted *Calluna vulgaris* at CO was less than half that of MT after three years. This is most likely a result of direct grazing effects and damage and uprooting due to trampling, corroborated by decreases in the height, width and length of the surviving cuttings at CO. Shrubs are more sensitive to herbivory than graminoids (Pollock *et al.* 2007), and therefore planting *C. vulgaris* at grazed peat bog sites does not aid restoration. Although the cuttings had much greater growth and survival rates at MT, regeneration from the heather brash also produced notable shrub cover and vigorous individuals as large as those that were planted. There was no difference in vegetation cover between planted and unplanted plots in 2019, indicating that the effort involved in harvesting and utilising cuttings does not provide any additional benefit to restoration in the absence of grazing if sufficient levels of heather brash are applied. Even though *C. vulgaris* has been used extensively in moorland restoration, it should be emphasised that it is not the main component of blanket mire vegetation (Proctor *et al.* 2013). Instead, a long term aim of peatland restoration is usually re-establishing *Sphagnum* (Rochefort 2000, Lunt *et al.* 2010). Planted or seeded heather can provide a key stepping stone between a nurse crop for peat stabilisation and more typical mire vegetation (Proctor *et al.* 2013), but its complete dominance would not be a desirable outcome. A dense, closed heather canopy on blanket peat will have a lower biodiversity value than a more species-rich community mosaic, and may indirectly increase atmospheric CO_2 emissions if its presence causes greater rates of decay in the soil (Dixon *et al.* 2015). Ferland & Rochefort (1997) found that ericaceous shrubs were not effective companion plants for facilitating *Sphagnum* establishment. In suboptimal conditions *C. vulgaris* may even shade out *Sphagnum* or lead to its loss through the accumulation of standing dead material in the absence of grazing (Lunt *et al.* 2010). The distribution of *C. vulgaris* in the eroded landscape is influenced by elevation, and it is most abundant on drier parts of the microtopography, for example hummocks and high lawns (Laine *et al.* 2007, Harris & Baird 2019). Any planting of this species would be better focused on these locations, rather than directly into bare peat. Alternatively, planting at restoration sites could incorporate more appropriate nurse plants such as *Eriophorum angustifolium*, an early coloniser of eroded bare peat and flat gullied regions which acts as a precursor to a more diverse community including *Sphagnum* (Ferland & Rochefort 1997, Boudreau & Rochefort 1998, Crowe *et al.* 2008, Harris & Baird 2019).

Long term implications

This study has shown that at MT in the absence of grazing impacts, significant revegetation of bare peat can be achieved in the four years following treatment with heather brash. Continued degradation of vegetation cover at the grazed site (CO) is anticipated due to the sustained effects of trampling. In addition, the newly established vegetation at MT is still fragile and vulnerable to pressures from frost heave, wind and rain. Over time the geotextiles are expected to biodegrade, and it is possible that areas of bare peat could reoccur in places with less stability (Anderson *et al.* 2011), particularly where thinner levels of brash were applied or the geotextiles were not used. Further applications of heather brash would be beneficial under this scenario. However, even if vegetation cover increases relatively quickly, long time scales are required to establish new steady states (Alderson *et al.* 2019). Peatlands can undergo extended periods of transition and readjustment following conservation interventions, particularly in terms of species composition (Taylor *et al.* 2019). A fully functioning blanket bog can take decades to form even when assisted by restoration management (Lucchese *et al.* 2010, Lunt *et al.* 2010, Lindsay *et al.* 2014b). The benefit of revegetation work will be maximised if it is carried out in conjunction with other proven restoration interventions such as reprofiling and ditch blocking, and takes into account the site specific geomorphological context of the erosion that is being repaired (Rochefort *et al.* 2003, Crowe *et al.* 2008).

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Appendix 1: List of blanket bog indicator species detailed in JNCC (2009); those highlighted in bold were recorded during this study.

Andromeda polifolia

Arctostaphylos spp.

Betula nana

Carex bigelowii

Calluna vulgaris

Cornus suecica

Drosera spp.

Erica spp.

Empetrum nigrum

Eriophorum angustifolium

Eriophorum vaginatum

Menyanthes trifoliata

Myrica gale

Narthecium ossifragum

non-crustose lichens

pleurocarpous mosses

Racomitrium lanuginosum

Rubus chamaemorus

Rhynchospora alba

***Sphagnum* spp.**

Trichophorum germanicum

***Vaccinium* spp.**

Appendix 2: List of all species recorded in the plots throughout the study arranged by species group (GR: graminoids, HB: herbs, SH: shrubs, BR: bryophytes, LI: lichens).

Species code	Species Name
GR	*Agrostis* spp.
GR	*Anthoxanthum odoratum*
GR	*Avenella flexuosa*
GR	*Carex echinata*
GR	*Carex pilulifera*
GR	*Deschampsia cespitosa*
GR	*Eriophorum angustifolium*
GR	*Eriophorum vaginatum*
GR	*Festuca ovina/vivipara*
GR	*Juncus articulatus*

GR	*Juncus squarrosus*
GR	*Luzula multiflora*
GR	*Luzula campestris*
HB	*Cerastium fontanum*
HB	*Equisetum palustre*
HB	*Galium saxatile*
HB	*Potentilla erecta*
HB	*Viola palustris*
SH	*Calluna vulgaris*
SH	*Empetrum nigrum*
SH	*Vaccinium myrtillus*
BR	*Aulacomnium palustre*
BR	*Bryum pseudotriquetrum*
BR	*Campylopus flexuosus*
BR	*Dicranum scoparium*
BR	*Hylocomium splendens*
BR	*Lophozia ventricosa*
BR	*Plagiothecium undulatum*
BR	*Pleurozium schreberi*
BR	*Polytrichum commune*
BR	*Polytrichum strictum*
BR	*Ptilidium ciliare*
BR	*Racomitrium lanuginosum*
BR	*Rhytidiadelphus loreus*
BR	*Rhytidiadelphus squarrosus*
BR	*Sphagnum* spp.
BR	*Tetraplodon mnioides*
BR	*Thuidium tamariscinum*
LI	*Cladonia arbuscula*
LI	*Cladonia bellidiflora*
LI	*Cladonia crispata*
LI	*Cladonia chlorophaea* agg.
LI	*Cladonia diversa*
LI	*Cladonia floerkeana*
LI	*Cladonia furcata*
LI	*Cladonia gracilis*
LI	*Cladonia portentosa*
LI	*Cladonia ramulosa*
LI	*Cladonia rangiferina*
LI	*Cladonia uncialis*
LI	*Hypogymnia physodes*