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

S.H. Watts

Biological and Environmental Sciences, School of Natural Sciences, University of Stirling, UK

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 CO₂-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 CO₂ 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 I 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 km² 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 m³ bag of brash covering an area of approximately 50 m². 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 m². 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.

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

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.
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: 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 1](https://www.mires-and-peat.net/)
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).

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

International Mire Conservation Group and International Peatland Society, DOI: 10.19189/MaP.2019.AJB.StA.1902
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.

<table>
<thead>
<tr>
<th>Damage type</th>
<th>CO 1</th>
<th>CO 2</th>
<th>CO 3</th>
<th>CO 4</th>
<th>MT 1</th>
<th>MT 2</th>
<th>MT 3</th>
<th>MT 4</th>
</tr>
</thead>
<tbody>
<tr>
<td>&gt;25 hoof marks</td>
<td>12</td>
<td>15</td>
<td>15</td>
<td>15</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>&gt;50 % netting unravelling</td>
<td>2 J</td>
<td>4 J</td>
<td>5 J</td>
<td>5 J</td>
<td>0</td>
<td>0</td>
<td>1 J</td>
<td>1 J</td>
</tr>
<tr>
<td>&gt;10 % netting pulled off</td>
<td>3 C</td>
<td>3 C</td>
<td>2 C</td>
<td>1 C</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>&gt;50 % netting pulled off</td>
<td>2 C</td>
<td>2 C</td>
<td>3 C</td>
<td>4 C</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>

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 ($\chi^2 = 14.90$, $P < 0.001$). 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 peat bog 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
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 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.
Peat bog 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 x 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
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 & Rochefort 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 enclosure 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). 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 tericolous 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 CO₂ 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 (Lucches e 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 reprioritising 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).

**ACKNOWLEDGMENTS**

The restoration work at Ben Lawers was funded by the Green Stimulus Peatland Restoration Fund through Scottish Natural Heritage (now NatureScot). Ben Lawers property staff (National Trust for Scotland) planned and implemented the project, including bagging and spreading heather brash and geotextile laying. Anthony McCluskey assisted with setting up the study plots in 2015 and Violaine Aurel assisted with the data collection in 2016. Many thanks to two reviewers for suggesting improvements to the manuscript.

**REFERENCES**


Borkenhagen, A.K., Cooper, D.J. (2019) Establishing
Borkenhagen, A., Cooper, D.J. (2016) Creating fen
Bain, C.G., Bonn, A., Stoneman, R., Chapman, S.,
Armstrong, A., Holden, J., Kay, P., Foulger, M.,
Atherton, I., Bosanquet, S.D., Lawley, M. (eds.)
International Mire Conservation Group and International Peatland Society, DOI: 10.19189/MaP.2019.AJB.StA.1902
Nature of Farming: Low Intensity Farming in
Nine European Countries
Deer Management Group, Aberfeldy, UK, 64 pp.
Background Information & Policies.
Noort, R., Wilson, J.D., Worrall, F. (2011)
Thompson, D.B.A., Thompson, P.S., Van de
A., Orr, H., Reed, M., Smith, P., Swales, V.,
R., Littlewood, N., Lunt, P., Miller, C.J., Moxey,
A., Orr, H., Reed, M., Smith, P., Swales, V.,
Thompson, D.B.A., Thompson, P.S., Van de
Noort, R., Wilson, J.D., Worrall, F. (2011) IUCN
UK Commission of Inquiry on Peatlands. IUCN
Background Information & Policies. Breadalbane
Deer Management Group, Aberfeldy, UK, 64 pp.
Beaufoy, G., Balduck, D., Clark, J. (1994) The Nature of Farming: Low Intensity Farming in
Borkenhagen, A., Cooper, D.J. (2016) Creating fen


Lindsay, R., Birnie, R., Clough, J. (2014a) *Briefing Note No. 7: Grazing and Trampling*. IUCN UK Committee Peatland Programme, Edinburgh, UK, 1 p.

Lindsay, R., Birnie, R., Clough, J. (2014b) *Briefing Note No. 9: Weathering, Erosion and Mass Movement of Blanket Bog*. IUCN UK Committee
Peatland Programme, Edinburgh, UK, 4 pp.


Submitted 26 Nov 2019, final revision 02 Jly 2020
Editor: Andy Baird

Author for correspondence: Sarah Watts, Biological and Environmental Sciences, School of Natural Sciences, University of Stirling, Stirling, FK9 4LA, UK. Email: s.h.watts@stir.ac.uk

S.H. Watts  REVEGETATION OF ERODED BARE PEAT IN REMOTE UPLANDS
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).

<table>
<thead>
<tr>
<th>Species code</th>
<th>Species Name</th>
</tr>
</thead>
<tbody>
<tr>
<td>GR</td>
<td>Agrostis spp.</td>
</tr>
<tr>
<td>GR</td>
<td>Anthoxanthum odoratum</td>
</tr>
<tr>
<td>GR</td>
<td>Avenella flexuosa</td>
</tr>
<tr>
<td>GR</td>
<td>Carex echinata</td>
</tr>
<tr>
<td>GR</td>
<td>Carex pilulifera</td>
</tr>
<tr>
<td>GR</td>
<td>Deschampsia cespitosa</td>
</tr>
<tr>
<td>GR</td>
<td>Eriophorum angustifolium</td>
</tr>
<tr>
<td>GR</td>
<td>Eriophorum vaginatum</td>
</tr>
<tr>
<td>GR</td>
<td>Festuca ovina/vivipara</td>
</tr>
<tr>
<td>GR</td>
<td>Juncus articulatus</td>
</tr>
</tbody>
</table>
Juncus squarrosus
Luzula multiflora
Luzula campestris
Cerastium fontanum
Equisetum palustre
Galium saxatile
Potentilla erecta
Viola palustris
Calluna vulgaris
Empetrum nigrum
Vaccinium myrtillus
Aulacomnium palustre
Bryum pseudorriquetrum
Campylopus flexuosus
Dicranum scoparium
Hylocomium splendens
Lophozia ventricosa
Plagiothecium undulatum
Pleurozium schreberi
Polytrichum commune
Polytrichum strictum
Ptilidium ciliare
Racomitrium lanuginosum
Rhytidiadelphus loreus
Rhytidiadelphus squarrosus
Sphagnum spp.
Tetraplodon mnioides
Thuidium tamariscinum
Cladonia arbuscula
Cladonia bellidiflora
Cladonia crispata
Cladonia chlorophaea agg.
Cladonia diversa
Cladonia floerkeana
Cladonia furcata
Cladonia gracilis
Cladonia portentosa
Cladonia ramulosa
Cladonia rangiferina
Cladonia uncialis
Hypogymnia physodes