Annex A

Restoration of blanket bog vegetation for biodiversity, carbon sequestration and water regulation

As part of Defra project BD5104

Literature review on:

‘Potential techniques to address heather dominance and help support appropriate ‘active’ *Sphagnum* supporting peatland vegetation on blanket bog and identify practical management options for experimental testing’

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1. Background and objectives

Uplands cover over 25% of the UK (Haines-Young et al., 2000). The bulk of this area comprises blanket bog, dwarf-shrub heath and acid grassland, and about 5 - 15% is managed for red grouse (Grant et al., 2012). The term "blanket bog" is often used rather loosely and falls within the overarching term of wetland, where conditions are such that soil water-logging favours *Sphagnum* moss growth and peat formation such that bog development leads to a smothering mantle on all but the more steeply sloping ground with an average thickness of 0.5-3.0 m (JNCC, 1999). Importantly, the UK contains about 15% of the global blanket bog areas (Evans et al., 2006), containing many specialist species of birds and plants. Moreover, the terms "mire" and "bog" are often seen as inter-changeable within the context of UK blanket bog and are in fact complex ecosystems with quite different hydrology, soil structure and plant species composition (O'Brien et al., 2007). During the last 25 years a significant proportion of blanket bog in the UK has undergone a programme of restoration. However, most of this work focused on degraded or eroded areas (mainly with grip blocking and/or re-vegetation) with little science-based evidence on implications for ecosystem processes (O'Brien et al., 2007). Even less is known about the implications of current or potential alternative management options for the heather dominated seemingly 'intact' blanket bog areas on which this project focuses. The question of "restoration of, and to, what" remains largely unanswered as most blanket bogs are strongly human influenced environments; whereas it may be possible to reverse a trend, it seems unlikely that such areas could be restored to their original state (Bradshaw, 2002). Most upland areas have been deforested by human activity since Neolithic times by fire and grazing (Tucker, 2003) and at least in Scotland the term 'muirburn', and thus management practice, goes back to medieval times (see Worrall et al., 2010b). This report focuses on the restoration aim of supporting a hydrologically functioning 'active' peat body, as it is this function which is pivotal to restoring the hydrological integrity, which could be hindered by ling heather (*Calluna vulgaris*) dominance. Crucially, 'active' blanket bog (supporting a significant area of vegetation that is normally peat forming; see JNCC, 2006) is protected under the EC Habitats Directive as a priority habitat. As recommended by O'Brien et al. (2007), sustainable catchment management (including potential changes to prescribed burning and grazing) requires further research with replicated long-term studies at the catchment and plot-scale to increase our understanding of the hydrological and hydrochemical processes. In fact, in the UK, many areas that were once 'active' blanket bog now support vegetation with extensive stands of *Calluna,* and may not be treated as blanket bog by all concerned. It is therefore crucial to understand both the processes that lead to such a change and how to restore the hydrological properties and thus achieve sustainable upland catchment management.

In view of this evidence gap, Defra commissioned a research project (BD5104) to use a rigorous experimental design to assess the alternative management options for heather dominated blanket bog vegetation to restore biodiversity, carbon sequestration and water regulation. As part of this research project, this literature review aimed to identify suitable experimental methods to reduce ling heather (*Calluna vulgaris*) 'over'-domination of blanket bogs in favour of 'active' peatland forming vegetation, especially with regard to the restoration of hydrological function and *Sphagnum* growth. *Sphagnum* is recognised as a unique global carbon store, containing more carbon than any other plant genus (Clymo, 1997), and is present at an abundance of over 80% in blanket bogs (Gunnarsson, 2005). However, despite this important role little is known about its dispersal and establishment potential. Peatlands (including the habitats bogs and dwarf shrub and heath) deliver a wide range of ecosystem services (Haines-Young & Potschin, 2008) that contribute to human well-being, including climate regulation, water purification, maintenance of biodiversity, recreational and educational opportunities, and tourism (Kimmel and Mander, 2010). Particularly in the UK, blanket bogs are of significant importance for drinking water supplies. Therefore, this review also considered how the identified management methods might affect ecosystem services, focussing on biodiversity, carbon sequestration and greenhouse gas regulation, as well as water regulation and quality. Thus, it provided the context for the planned experimental work both at plot and catchment level scales within the overall Defra project (BD 5104). Furthermore, grip-blocking is recognised as an effective method of raising the water table (Holden, 2009b), which will likely continue to be carried out extensively across the UK, especially in England and Wales, over the next 5-10 years. **Therefore, the aim of this review was the consideration of supplementary land-management methods (i.e. methods in addition to grip blocking, where applicable) that could be used to address heather domination whilst also minimising adverse impacts on ecosystem services, including increased methane (CH_4) emissions and dissolved or particulate organic carbon (DOC; POC) losses from restored areas.** Throughout the report related knowledge gaps are identified and highlighted in the corresponding sections. Rather than repeating more general information and description on carbon sequestration and hydrological concepts, the reader is referred to another technical report to Defra (BD1241) by O'Brien et al. (2007).

Although moorland drainage has been practised in the past, and continues to be the case in some places, the main agricultural and sporting related management practices currently conducted in moorland areas are grazing, burning and cutting, and predator control (Backshall et al., 2001). In the UK, grouse shooting estates cover an estimated area of between 0.66 to 1.7 million hectares (Grant et al., 2012). Prescribed burning of *Calluna* has been used traditionally to maximise grouse densities through preventing establishment of woody species and encouraging nutrient cycling, thus stimulating *Calluna* growth and dominance, and to create a mosaic of different aged stands to provide both food from new young growth and protection from predators in the taller, older stands. In the UK, about 18% of peatlands (Worrall et al., 2010b) and 30% of blanket bog (Natural England, 2010) are estimated to be under a burn rotation. Crucially, dominance of *Calluna* at the plot-scale has been shown to impede *Sphagnum* growth due to increased evapotranspiration (see Worrall et al., 2010b) and thus lowered water table depth (Worrall et al., 2007; Lindsay, 2010). *Sphagnum* mosses have no vascular system and as such may disappear or not establish if mean water table depths falls 20 cm below the peat surface (Ivanov, 1981). However, heather does not necessarily suppress *Sphagnum* species, as heather also offers protection for *Sphagnum* growth, mainly in older (rank) stands of *Calluna*; in many cases it is difficult to know if heather itself or management (i.e. burn) is mainly responsible for the lack of 'active' bog species. Although burning mostly occurs on shallow peat and podzolic soils, it regularly encroaches on deeper blanket bog peat areas (Yallop et al., 2005), and often peat depth is unknown to, or not necessarily taken into account by, the land manager. Furthermore, it can be assumed that many areas of shallow peat once supported deeper peat and it is likely that land management, such as over-grazing or burning, stimulated decomposition and erosion processes, possibly reducing peat depth in addition to any losses from peat cutting (Ardron, 1977). However, this hypothesis requires further research. Notwithstanding this uncertainty, overgrazing, burning and drainage are seen as the top three reasons for adverse conditions of Sites of Special Scientific Interest (SSSI) blanket bog areas (Natural England, 2008), of which only around 12% are in favourable condition (Natural England, 2015).

Nevertheless, there are strong differences of opinion and it has been suggested that the parties involved (e.g. government agencies, landusers and other stakeholders) need to reach a common science-based understanding on the impacts of burning on moorland (EFRA Committee, 2004). For example, Di Folco and Kirkpatrick (2011) showed soil carbon (C) losses in Tasmania from planned fires on moorland, Clay et al. (2010) highlighted the importance of burn rotation length for long-term carbon budget. Moreover, Worrall et al. (2010b) concluded that the evidence for the effects of burning on peatlands on ecosystem services was equivocal, although on balance it indicated several negative impacts such as on water quality. This project acknowledges that the term 'over dominance' of heather is difficult to interpret, clearly depending on one's view point; here the term is mainly used to refer to a lack of 'active' bog species. **Rather than defining the term 'over-dominance', the review's focus is on summarising the concepts, and acquiring the scientific evidence and data, to determine what the likely impacts of different management options might be on plant biodiversity, carbon sequestration and water regulation.** Clearly much of the science on the wider environmental impacts is still at an early stage and a considerable research effort is needed to understand how different blanket bog management practices affect C storage and export, and water flow and quality, via changes in vegetation cover.

2. Management impacts

Introduction

This review assesses current knowledge of management options with respect to restoration of 'active' blanket bog, particularly regarding the restoration of hydrological function and *Sphagnum* growth, the focus being on those options of most relevance to the management of heather dominated blanket bog vegetation:- grazing (Section 2.1), burning (Section 2.2) and cutting (Section 2.3). Herbicides (Section 2.4) have also been used as part of moorland restoration programmes (Backshall et al., 2001), but mainly for the control of bracken and to a more limited extent *Molinia*. This report considers these four management options and highlights their relevance to supporting 'active' bog vegetation and their potential impacts on key ecosystem services of relevance to this project. However, due to their potential environmental and ecological impacts (located often within SSSIs), herbicides are generally seen as less appropriate to blanket bog management in this context. Other potential management options are briefly considered in Section 2.5. The penultimate part of the review (Section 3) covers possible effects on other ecosystem services, such as greenhouse gas (GHG) emissions, although erosion is largely discussed in Section 2.2.1 (burning). The report concludes with recommendations for experimental treatments (Section 4) to be applied as part of this Defra project (BD5104).

2.1. Grazing

Grazing of domestic livestock has been an important component of land use in the uplands of Britain. In fact, historically it can be argued that the present state of blanket bog is undeniably related to such management, thereby creating a semi-natural system (Moore, 1993; Tallis, 1998). Traditionally, low-intensity summer grazing had little effect on vegetation composition and might have even helped to maintain the balance of species present in bog communities (O'Brien et al., 2007). With the introduction of government support schemes, such as the former Hill Livestock Compensation Allowance (HLCA) based on a headage payment, sheep numbers have increased on many areas of blanket mire (ADAS, 2004). This increase, combined with the loss of 'traditional' management practices such as shepherding, has resulted in many upland peatlands becoming overgrazed, leading to changes in the vegetation, and erosion and localised compaction of the peat. However, there are other natural and anthropogenic triggers and drivers of erosion (e.g. climate, burning, draining), and their contributions in most cases are quite unclear (Lindsay, 2010). Although grazing intensity has since been reduced in many areas, Natural England (2010) estimate that 5% of all deep peats are still in this undesirable state (9% on upland blanket bog); based on a simplified version of the Durham Carbon Model, Natural England (2010) estimate that overgrazed blanket bogs release CO_2-equivalents (i.e. including methane and nitrous oxide emissions) of 0.1 tonnes CO_2-e ha^{-1} yr^{-1} compared with a net uptake of 4.11 tonnes CO_2-e ha^{-1} yr^{-1} by undamaged blanket bogs.

Blanket bogs are sensitive to overgrazing because they are slow-developing habitats**.** Too much grazing, and the associated trampling and uprooting, can lead to the loss of slow growing heather and especially peat forming moss species (English Nature, 1996), whilst increasing cotton grass species and other graminoids (especially *Molinia* and *Trichophorum*) and/or leading to an increase in bare ground and erosion (Martin et al., 2013). On the other hand, light year-round grazing may weaken heather domination and increase the structural diversity of the vegetation (Rawes and Heal, 1978 in English Nature, 1996), which in turn can increase invertebrate diversity (ADAS, 2004). Even when stocking rates are low, there may be instances of local high grazing levels. The intensity and extent of grazing impact varies according to many factors, is complex and justifies further long-term research (see O'Brien et al., 2007). Generally, the damage that can be caused by trampling of vegetation (making ground soft or muddy) depends on the stocking density but will be concentrated around fence lines and preferred feeding grounds with larger animals such as cattle causing more damage than smaller ones such as sheep (Shaw et al., 1996). Wet areas are usually more susceptible to trampling, and thus soil structural damage, than dry ones -

the margins of pools can be severely damaged, leading to loss of *Sphagnum* mosses and development of areas of bare peat (RSPB, 1995 in Shaw et al., 1996; Backshall et al., 2001). *Sphagnum* species are particularly susceptible to trampling, which may thus cause changes in microtopography and species distribution, for example a reduction in hummock-forming species such as *S. papillosum* or *S. palustre*. However, *S. tenellum* may benefit, as it commonly colonises wet bare peat and is often associated with compacted areas and animal tracks (Carroll et al., 2009).

Recommendations on stocking densities are confusing and require more research (O'Brien et al., 2007); optimum stocking rates and breeds of sheep and their effects on hydrological processes require long-term research, and most studies on grazing impacts were conducted on dry heath (Shaw et al., 1996). There are no agreed stocking densities for blanket bog *per se*, and these could be seen as a 'blunt instrument', as one density would unlikely to be appropriate on all sites (for a review see Glaves, 2008). Mostly, studies on grazing impacts are aimed at assessing effects on erosion (Evans, 1997) or shifts to grassland (Thompson et al., 1995). Negative effects on heather seem to increase at stocking densities of <0.4 ha per sheep (>2.5 sheep ha^{-1}) (Grant et al., 1985), well within the range likely to cause erosion effects (O'Brien et al., 2007). Disruption of the physical structure of the surface peat can result in wastage and oxidation of the peat and can presumably also increase the risk of erosion (see Sections 2.1 and 3a). In areas supporting mosaics of different vegetation types, native grass communities are likely to attract a higher grazing pressure throughout the year than blanket bog or wet heath communities, and shepherding may be required to maintain a more even grazing pressure (Shaw et al., 1996). Sheep can also preferentially graze sensitive areas, such as 'sweet' mineral flushes, to the detriment of such flushes. On grouse moors and other burned moorland, distribution of grazing pressure will be influenced by rotational muirburn, which can act to distribute sheep and deer more evenly across a moor, because of the resultant distribution of nutritious new growth from rotational burning (Grant et al., 2012).

There is contradictory evidence on the relationship between grazing and enrichment of the substratum. Supplementary feeding may introduce more nutrients into the system and, unless the animals are taken off the site overnight, defecation and urination can raise the soil's nutrient status; although the effect is likely to be localised (RSPB, 1995 in Shaw at al., 1996). This increased nutrient input (likely benefitting fast growing species more than slow growing bog species) needs to be considered as part of any grazing scheme. According to Harrison (1985) and Marrs et al. (1989), in the long term grazing may lead to a reduction in soil fertility as long as nutrients are either locked up in the peat or leave the system in the animals; however, experimental evidence from Moor House from up to 31 years of grazing exclusion experiments does not support this hypothesis (Marrs et al., 1989). Rawes and Heal (1978) discussed the nutrient inter-relationships between different habitats available for sheep grazing at Moor House National Nature Reserve (including blanket bog) and concluded that there was little or no net income to or loss from the bog of N, P, K and Ca as a result. However, deposition of dung can be beneficial in increasing the invertebrate food supply for some birds.

Sheep and cattle are the main grazing animals on moorlands, with some horses, ponies and feral goats. Grazing animals other than domestic stock include deer and rabbits (Backshall et al., 2001). The overall effect of grazing on bog vegetation is to reduce the quantity of more palatable plant species and thus increase the frequency of less palatable and more resistant species (e.g. grasses, low and rosette-forming species and distasteful plants). The specific effects of grazing on peat soils are complex and will depend on a number of factors, including the following (Ausden and Treweek, 1995 cit. in Backshall et al., 2001): the types of vegetation present; the condition of the vegetation (including the proportion of different age classes of heather, if present); the timing of grazing (whether year round, seasonal or occasional); the intensity of grazing (numbers of animals, length of grazing period); the type of grazing animal (species, breed, age, sex); the associated practices (supplementary feeding, presence and nature of shepherding practices, use of vehicles); the other management practices conducted (burning, cutting, bracken management); the existence and location of fences and other boundaries; the geographical location of the area; the underlying geology, soil type and wetness of the area; and the climate, topography, altitude and aspect.

Heavy grazing on poorer soils can bring about the replacement of heather with swards dominated by unpalatable matgrass and heath rush. There is also evidence that heather can be checked, and bilberry encouraged, by certain grazing pressures (Welch et al., 1994). Moreover, in a 15-year sheep grazing exclosure experiment carried out on poor quality blanket bog (species poor, little cover, active erosion) at Moor House, major changes in species composition, pattern of vegetation and structure were observed, with *Calluna* growing, from an early position of scarcity, to dominate parts of each exclosure (Rawes, 1983). Hartley and Mitchell (2005) found a decrease in *Calluna* cover under grazing with N fertilisation, but not when grazing was excluded, with N fertilisation generally decreasing bryophyte cover.

Backshall et al. (2001) provide guidelines, including recommended stocking rates, for the sustainable grazing of blanket mire; generally, the wetter the site the lower the productivity of the blanket mire plants and the greater the sensitivity to grazing (Coulson et al., 1992). Hare's-tail cotton-grass (*Eriophorum vaginatum*) and heather form the majority of the sheep diet on blanket mire during the winter (Grant et al., 1976), during which *Calluna* is particularly susceptible to grazing (Grant et al., 1987). Cotton grass (*Eriophorum* spp.) is also favoured in early spring, when the flower stems and leaf bases emerge from the dead tussocks (Mowforth and Sydes, 1989). It is also grazed in late summer in preference to heather when the productivity of the more palatable grasses falls (Rawes and Williams, 1973). Light summer grazing can help to reduce heather domination, shrub invasion and cotton-grass competition in recovering situations. High levels of grazing can reduce the cover of *Sphagnum* and lichens, while cotton-grass species and/or purple moor-grass (*Molinia caerulea*) increase in dominance (Coulson et al., 1992; Rawes and Hobbs, 1979). Light grazing by sheep, without burning, is likely to be an acceptable management for blanket bogs in the interests of conservation (Rawes and Hobbs, 1979). Where sheep grazing intensity was increased on blanket bog either experimentally (Rawes and Williams, 1973) or through long-term management (Welch and Rawes, 1966), the major changes were a decline in the standing crop of heather and an increase in hare's-tail cotton-grass. The presence of heath rush (*Juncu*s spp.) in more intensely grazed bogs (0.75 or more sheep ha^{-1}) indicates that *Juncus*-dominated swards are at least partially caused by heavy grazing pressure (Backshall et al., 2001).

However, heavy grazing is always likely to eventually cause erosion and other related negative impacts on ecosystem services; according to DARDNI (2011), overgrazing of heather moorland can be identified by:

- heather cover restricted to small clumps between mainly grassy areas;
- overgrazed heather plants with distinctive domed shapes;
- areas of bare ground and sheep tracks through the area along with high concentrations of dung, impaired drainage and the likelihood of standing surface water;
- severe soil structural damage, especially in wet weather, exposing bare peat;
- the gradual disappearance of heather cover over time and a change to less productive grasses.

Heather moorland is particularly vulnerable to livestock damage during the winter. Heather needs a rest period to build up plant reserves and this does not occur under continuous grazing (DARDNI, 2011). Winter grazing also destroys next season's growing points and can lead to trampling and soil structural damage. Commonly, to ensure good heather growth, stock should be removed from heather moorland between 1 November and 28/29 February and stock levels controlled during the grazing season. Light autumn/winter grazing during establishment phases and summer grazing during regrowth could be considered as ways of weakening heather over time, thus providing indirect support for slow growing 'active' bog species. For sheep grazing all year round, Hulme and Birnie (1997) calculated that grazing levels of between 0.5 and 1 sheep ha^{-1} are the maximum level allowable if blanket bog vegetation is not to suffer damage. Lindsay (2010) notes that their calculation is based on a number of simplistic assumptions and so it would be better to remain on the side of caution, the lower limit of their range being quite similar to the figure obtained by Rawes and Hobbs (1979) for the grazing tolerance of vegetation at Moor House NNR. Similarly, the maximum stocking rates recommended by DARDNI (2011) for blanket bog are 0.5 ewes ha^{-1} and they do not recommended grazing cattle on blanket bog.

Knowledge of the effects of grazing on moorland bird populations is extremely poor (Fuller, 1996 cited in Backshall et al., 2001) although, in general, low levels of grazing are most appropriate for the conservation of moorland bird populations including most ground-nesting waders (Backshall et al., 2001). A good diversity of vegetation structure is important for both invertebrates and small mammals and this can also be maintained by light grazing. On the other hand, overgrazing can produce a more uniform structure which is generally poor for invertebrates and can lead to the loss of small mammals as well as the species dependent upon them (Backshall et al., 2001).

There are very few full carbon or greenhouse gas (GHG) budget studies for peat soils in the UK, and even fewer that consider land management and changes in vegetation composition (Worrall et al., 2011). Impacts on peatland C dynamics of light grazing do not appear to be as great as for burning. Although grazing reduced above-ground C stocks and increased gross ecosystem CO_2 fluxes of respiration and photosynthesis, it had no significant effect on C stocks in surface peat (Ward et al., 2007). Garnett et al. (2000) found that light sheep grazing did not affect rates of peat C accumulation during a long-term, randomised block experiment on blanket bog at Moor House in the North Pennines. However, any soil carbon stock changes are difficult to detect within 10-20 years, particularly in C rich peatlands where the measurement error and detection limits are considerable (Rodeghiero et al., 2009). Regardless of possible soil C stock changes, there are likely to be changes in GHG emissions related to changes in plant functional types (PFTs). Sedges and rushes are known to cause plant mediated transfer of CH_4, even under lower water table depths, due to their deep aerenchyma roots. On the other hand, heather tends to dry out peat soils due to its high transpiration rates, thus leading to less CH_4 production and release (i.e. heather roots do not have aerenchyma root) and is associated with the development of 'peat pipes' (Holden, 2005) which may affect hydrology and DOC loss. However, more research on PFT impacts on carbon and water cycling is needed under controlled experimental conditions.

2.2. Burning

Burning has been used to manage upland vegetation in Britain for centuries, principally for stimulating new growth of grasses or heather. However, large scale and repeated heather burn rotations (typically 10% in patches of about 0.5 – 1.0 ha in size on a 10-15 year rotation) have only been introduced across the UK during the past 200 years (e.g. Hay, 2012) in relation to grouse management. However, burn rotations vary regionally and shorter rotations have been noted (Yallop et al., 2006). Careful, periodic burning of upland vegetation can have advantages for agriculture, game rearing, wildlife conservation and intrinsic landscape appeal. However, inappropriate and uncontrolled/out of control fires in the uplands can be more damaging than a complete lack of burning management, although the latter poses the risk of establishing a build-up of biomass fuel loads (Clay et al., 2010b). The most commonly burnt upland vegetation type is dwarf shrub heath, although some burning of intact blanket bog is also undertaken. Burning alters the vegetation composition, pattern, physical and age structure, nutrient status and carrying capacity for herbivores, as well as the associated fauna. Some 15% of all deep peats in England are subject to rotational burning of heather (also called rotational muirburn in Scotland, going back to at least medieval times (Worrall et al., 2010b)), including 30% of blanket bog (Natural England, 2010). This management encourages different age structures of heather to support high grouse populations, but when done poorly or when fires get out of control, it can also kill or damage the peat and bog mosses mainly responsible for the formation of peat (Natural England, 2010). Overall, muirburn effects on vegetation depend on many factors but, in general, seem to increase species diversity and richness on dwarf shrub heath (Grant et al., 2012); however, these findings might not apply to grouse moor managed areas, and few studies have assessed effects on blanket bogs (Grant et al., 2012). Overall, relatively little is known about effects of burn management on grouse moors on plant species (particularly peat forming ones), and effects of fire intensities and frequencies require more long-term research, particularly on blanket bogs (see e.g. Worrall et al., 2010b).

Burning management of any kind may be detrimental to blanket bog but the most severe damage is likely to occur when uncontrolled fires occur, which a careful regular burning regime might prevent (Yallop et al., 2006). This is because the intensity of a fire is determined by the temperature reached and speed of travel (Shaw et al., 1996). A slow-moving, hot fire will kill most, or all, of the plants (including bryophytes), so that there is no regeneration from stools and exposure of bare ground is prolonged. Conversely, most plants will survive a quicker-moving, cooler fire, particularly where the ground is wet, and, provided that there is no thick layer of litter, new shoots will rapidly appear. However, variability in fire intensity is high in general and not all wildfires are necessarily more damaging (Clay et al., 2010a). The initial vegetation composition (and hence the previous management history) can influence the fire intensity, and in old stands of heather fire may be particularly intense. An intense fire which causes ignition of the peat and subsequent erosion may cause major losses of soil organic matter and nutrients and lead to poor recovery of the vegetation (Shaw et al., 1996). Accidental summer fires are most likely to be of high intensity (although not necessarily so as they also show a high variability as do prescribed burns), and thus are likely to be particularly detrimental as (a) they are largely uncontrolled and may burn for long periods and over large areas; and (b) the peat/humus layer is more likely to be ignited (Hobbs and Gimingham, 1987). On the other hand, in summer, the vegetation itself, especially heather, tends to have a higher moisture content, and so less easily ignites.

Whether or not burning is appropriate for a piece of land will depend on the objectives for that particular area (e.g. for nature conservation, game, agriculture or landscape). Maintaining some areas as unburnt or burnt on a long rotation can be of great benefit for wildlife. However, large stands of woody heather (i.e. rank heather) or extensive areas of dense grass litter may develop in the absence of burning and grazing, and these can then pose a fire hazard. In situations where scope for alternative management is limited, controlled and careful burning may be appropriate to reduce the fuel load available to burn, to create fire breaks and generally reduce the likelihood of uncontrolled fires. Of particular consideration is the ground condition (e.g. vegetation cover, slope) in relation to regeneration and potential erosion. **Box 1** summarises the advantages and disadvantages of burning on blanket bogs.

Burning can maintain heather in the building stage of growth and prevent it from reaching the mature/degenerative phases. However, heather growth overall is faster on dry than on wet ground (Forrest and Smith, 1975 cited in Grant et al., 2012). Conversely, under certain conditions, frequent burning can eventually eradicate heather especially on blanket bog (see **Box 2** for a summary of all burning effects on vegetation). Shorter rotation burn cycles can cause a shift from *Calluna* to *Eriophorum* dominated blanket bog (Hobbs et al., 1984), and the balance between vascular plant species can clearly be altered by burning. Some plants have adaptations which render them relatively resistant to fire, e.g. growing points at or below ground level; purple moor-grass, deer grass (*Trichophorum cespitosum*) and hare's-tail cotton-grass are examples of such species (Rowell, 1988). Other species are more adversely affected by burning and changes in the competitive balance are as important as the effects on the vigour of particular plant species. Non-vascular plants can be harmed by burning, although there has been little research on this subject (Shaw et al., 1996). Moreover, seed longevity varies between species, and its long-lived seeds mean that *Calluna* will probably benefit from a long burn rotation, which can suppress germination of many other moorland species (Hobbs et al., 1984). The intensity (i.e. 'cool' *vs*. 'hot') and frequency of the fire (see **Box 3**) is critical. For example, damage to bryophytes can especially be avoided if a quick ('cool') fire occurs when the ground is frozen (Rowell, 1988). Some species, including certain, but not all, *Sphagnum* species, are also capable of rapid recolonisation following burning, at least in the lowlands (Daniels, 1991 cited in Backshall et al., 2001). Generally, low intensity fires on blanket bogs at more than 20-year intervals may have little long-term effect on *Sphagnum* species, but moderate to high intensity fires at return intervals of less than 20 years can eradicate them. However, results are conflicting, possibly reflecting specific ground and management conditions; one study (Hobbs, 1984) showed that, together with grazing, a short-term 10-year burning rotation was more beneficial to *Sphagnum* than a 20-year rotation.

Box 1: Advantages and disadvantages of burning on blanket mire (from ADAS, 2004)

Advantages

- Can stimulate heather seedling regeneration
- Increases structural diversity of sward – for example, by maintaining stands of heather in different growth phases, and opening up the sward to encourage a mosaic of short and tall vegetation which benefits curlew (*Numenius arquata*)
- A mosaic of heather at different growth stages benefits invertebrates e.g. Lepidoptera (Haysom & Coulson, 1998)
- The open habitat benefits some invertebrates
- Increased invertebrates increases the food availability for birds, small mammals, amphibians and reptiles
- Shorter vegetation is favoured by some birds e.g. golden plover (*Pluvias apricaria*)
- Strategic burning to encourage the growth of young heather, can be used to move sheep away from sensitive areas of bog vegetation
- Burning can be used to create firebreaks and to prevent the spread of accidental fire. This may be particularly necessary near footpaths.
- Where cotton grass (*Eriophorum* spp.) is sparse, burning to increase its frequency may benefit species which feed upon it, particularly rare species such as black grouse (*Tetrao tetrix*) and large heath butterfly (*Coenonympha tullia*) (Tucker, 2003)
- Can encourage cloudberry (*Rubus chamaemorus*) and bog rosemary (*Andromeda polifolia*) (Backshall et al., 2001)

Disadvantages

- May encourage competitive species such as purple moor grass (*Molinia caerulea*) and hare's-tail cotton grass (*Eriphorum vaginatum*)
- Can damage bryophytes particularly *Sphagnum* – *Sphagnum* spp. may survive low intensity fires but hot fires will be detrimental
- Can lead to bare ground which can increase desiccation and initiate erosion
- Can reduce the ability of heather to regenerate vegetatively by layering (Gimingham, 1988)
- Can reduce species richness of the vegetation
- Can result in an uncontrolled fire with larger patches of vegetation or peat being burned
- Can reduce diversity within the microtopography – areas subject to frequent or intense burning rarely display the complex array of hummocks, ridges and hollows typical of blanket mire
- Can draw in stock to burned areas and increase grazing/trampling pressure on the surrounding bog vegetation

The burning of blanket mire and wet heath is not required to maintain their nature conservation interest (Mowforth and Sydes, 1989; Rawes and Hobbs, 1979); rather, it is thought to reduce their conservation value (Usher and Thompson, 1993) and so should be minimised and, where possible, eliminated (Backshall et al., 2001). In an English Nature review of historical effects of burning and grazing (Shaw et al., 1996), it was stated that '*burning is not usually recommended for management of blanket bog although there may be a case for its infrequent use in some circumstances'*. This English Nature review concluded that regular burning regimes are likely to be damaging to the wildlife interests of blanket bog and wet heath, although if carried out sensitively, they can be advantageous to some species of these habitats. Shaw et al. (1996) also conclude that it is impossible to define one optimal management regime to cover all areas of blanket bog and upland wet heath throughout Britain due to local factors, past management regimes, precise management objectives and possible conflicts of interest. In a more recent English Nature review of burning in the uplands (Tucker, 2003) it was recommended that burning should not be carried out on blanket bog '*unless the conservation benefits are clear, or for the creation of firebreaks to prevent wildfire in high risk areas'*. This report concludes that further investigation of the impacts of the frequency and intensity of prescribed burning is required (see O'Brien et al., 2007). **Box 1** summarises in full the advantages and disadvantages of burning on blanket mire as they are listed in Defra's Upland Management Technical guideline No 4 (ADAS, 2004).

On a moor which is well managed by burning, moderate levels of sheep grazing can benefit both sheep and grouse by maintaining the heather in the more favoured pioneer and building phases. This has the added benefit of increasing the interval needed between fires, with concomitant reduction in the adverse impacts of too frequent burning (Shaw et al., 1996). Burning is thought to increase the nutritional value of heather to grazers by stimulating new growth, which has the highest concentrations of nutrients (Coulson et al., 1992). On the other hand, burning releases nutrients from plant material in smoke and ash, losses being greater following an autumn burn (as opposed to a spring burn) when fewer nutrients from the ash are taken up by the vegetation and more are lost through run-off and leaching (Shaw et al., 1996). Moreover, any stimulation of new growth requires nutrient replacement, which is likely to come from increased mineralisation of organic matter. Importantly, heather is a strongly mycorrhizal plant and their associated mycorrhizal ericoid fungi possess enzymatic qualities for acquiring nutrients from mineralisation of organic matter (potentially also including charcoal), which might therefore lead to slow but steady peat decomposition and thus carbon losses. For instance, larger DOC production was observed under heather canopies by Armstrong et al. (2012). More research on this possibility is needed (pers. comm. Jonathan Leake, University of Sheffield).

Box 2: Burning effects on blanket mire vegetation (from Backshall et al., 2001; chapter 6)

Burning has a marked effect both on the floristic composition and production of blanket mire and wet heath vegetation (Heal and Perkins, 1978; Backshall et al., 2001)). For example:

- unburnt bog can have greater species diversity than burned bog (Hobbs et al., 1984);
- some plants, notably *Sphagnum* mosses, can be eliminated by burning;
- hare's-tail cotton-grass recovers quickly after burning and can become dominant;
- hare's-tail cotton-grass above ground standing crop after five years can be about 65% of the total vascular plant community (Gore and Olson, 1967), and can assume permanent dominance if the community is burnt frequently (Rawes and Hobbs, 1979);
- a short burning rotation (every 10 years) can result in increased dominance by *Eriophorum* spp., while a long burning rotation can lead to greater abundance of heather after fire;
- crowberry, bilberry and grasses can be encouraged if burning rotations are short;
- heather regenerates more slowly, taking about 20 years to regain its full dominance, when it can contribute 70% to the above-ground standing crop (Forrest, 1971);
- heather on blanket bog may eventually be eliminated by a 10-year burning cycle;
- cloudberry and cross-leaved heath may dominate initially after fire, but are likely to be succeeded by heather during long intervals between fires (Mowforth and Sydes, 1989).

Notably, there is an on-going debate about the optimal management regime for the conservation of flora of blanket bog and wet heath, including whether or not to burn (Shaw et al., 1996; Stewart et al., 2004). In a review of primarily quantitative studies of the effects of burning on blanket bog or wet heath, Stewart et al., (2004) found no consistent significant changes in floristic composition in response to burning. Certain changes were interpreted as negative in the context of favourable condition, suggesting burning degrades blanket bog, although in one study the outcome was positive. Stewart et al. (2004) conclude that, pending further research, burning on blanket bog and wet heath should normally be avoided, although there is not a robust evidence-base to support management decisions regarding burning on blanket bog and a particular lack of long-term studies that deal with more than one burning rotation.

Although *Sphagnum* mosses (a key component of blanket bogs) are frequently considered to be susceptible to burning, the full effects of rotational muirburn on them, and how these effects vary according to differences in muirburn regimes, are poorly understood (Grant et al., 2012). For example (as cited in Grant et al., 2012), in some cases, burning may have little impact on *Sphagnum* (Daniels, 1991; Barkman, 1992) and MacDonald (2000) considers that *Sphagnum* may be relatively well able to survive “fairly low intensity” management fires vegetatively. Moreover, evidence from burning rotation experiments at Moor House NNR in the Northern Pennines (Lee et al., 2013) suggests that long burn rotations (>20 years) encourage *Calluna* domination. Especially in ‘unburnt’ (since ca. 1924) ‘reference plots’ the number of peat-forming species (*Eriophorum*/*Sphagnum*) did not increase, although bryophyte diversity but not cover increased (which was mainly *Hypnum jutlandicum*). Whereas shorter rotations (~10 years) encouraged *Eriophorum* dominance but with an increase in (the still low) frequency of *Sphagnum capillifolium* and bare ground (and notably, species frequencies were very low overall). In fact, vegetation height (>40 cm) (equivalent to >20 years since burning) seems to be an indicator for reduced plant diversity, as suggested by Harris et al. (2011) based on more severely modified, species-poor blanket bog in the Peak District; they recommend burning at vegetation heights above 25 cm to maintain the pre-burn complement of plant species. However, Grant et al. (2012) point out that there is a paucity of data on the effects on overall plant species diversity of rotational muirburn on blanket bog. Many studies describe the effects of burning on plant species diversity at the stand scale – comparing burnt and unburnt ‘patches’. However, when considering the overall effects of muirburn on plant species diversity, it is also necessary to consider effects at larger scales and account for the variation produced by creating a mosaic of different successional stages, and hence different species compositions. Therefore, if rotational muirburn at least maintains plant species diversity at the stand scale, then it should produce an overall increase in species diversity across a moor (Grant et al., 2012). This was illustrated for a dry heath by the Mar Lodge study of lichen species richness, in which both the total number of species present and species composition increased with time since burning (Davies and Legg, 2008), and increases in species richness were greater at the larger (e.g. moor or catchment) scale (Davies, 2001). However, the end point of the target diversity depends on the context and the target species, which in this study is a high diversity in peat-forming blanket bog species (i.e. not a heathland with likely higher overall diversity) and few data are available for blanket bog (Grant et al., 2012). Therefore, a recommendation for determining effects on plant diversity is to consider satellite or aerial pictures to assist with impact assessment at the larger catchment-scale, combined with long-term catchment-scale and plot-level studies.

By maintaining *Calluna* cover and creating areas of open vegetation, rotational muirburn helps to provide key habitat elements for some moorland bird species (e.g. nest-sites for some waders, notably golden plover, and suitable foraging habitat for several raptors), although burning may also have detrimental effects on the habitat conditions of other bird species such as the meadow pipit (Grant et al., 2012). The effects of grouse moor management on moorland invertebrates are poorly understood although a carefully managed, ‘sensitive’, rotational muirburn regime on blanket bogs is expected to increase the abundance of at least one species of conservation importance, the large heath butterfly (Grant et al, 2012). Interestingly, burning as a control method for heather beetle (*Lochmaea suturalis*) populations is sometimes still recommended, albeit with the acknowledgement that this needs to be a hot burn, damaging the litter and moss layer (where the adult beetles hibernate) and cutting might be as effective, but both possible control methods still need more research (Rosenburgh and Marrs, 2010). However, heather beetle grubs are the most vulnerable life stage and are not present during the legal burning period; hence, burning is unlikely to be an effective control of this pest (SEERAD, 2001).

During the last 20 years, a number of published guidelines have focussed on controlled burning on blanket bog, but these have given a mixed and confusing message (O'Brien et al., 2007). As a result of this confusion, and differences in opinion amongst upland land-based organisations and government agencies, Defra established a Burning Review Science Panel to review the impacts of prescribed burning on wet and dry upland heath and blanket bog (Glaves et al., 2005). The Panel concluded that, although there was a lack of science-based evidence, prescribed burning can be linked to degraded blanket bog. The Panel recommended that, in general, the presumption should continue to be that vegetation on blanket peat should not be burnt, unless as part of a restoration programme or to meet wider conservation/environmental objectives (Glaves et al., 2005). **Box 3** summarises recommendations concerning the burning of blanket bogs based on Backshall et al., 2001.

Box 3 Recommendations concerning the burning of blanket mire and wet heath

(Backshall et al., 2001; chapter 6)

- As a general rule when managing mires for nature conservation, if in doubt, do not burn (Brooks and Stoneman, 1997).
- Where blanket bog and wet heath is in favourable condition the ideal option for nature conservation purposes is not to burn at all.
- A 20-year burning regime is the recommended minimum rotation for blanket mires (Mowforth and Sydes, 1989) and a burning rotation of 20-30 years may be preferable.
- Where burning is conducted, for conservation purposes it is desirable to convert some areas (wetter, steeper, higher altitude locations) to no burning areas.
- When conducting any burning on blanket mire or wet heath, follow all the legal requirements, areas to be avoided and other recommendations (Defra, 2007).
- Large areas of old, tall heather on wet substrates are ideally left unburnt, because of the risk of very hot fires and little regeneration.
- Large areas dominated by cotton-grass (*Eriophorum* spp.) are best avoided because this will encourage these species, unless accompanied by stock reduction as part of a restoration phase.
- Areas which contain pools or peat hagging, and close to eroding gullies, should also not be burnt.
- Where accidental fires are likely and extensive areas of old, woody heather exist, burn fire breaks as a precaution (Mowforth and Sydes, 1989) or consider cutting fire breaks, but consider the possibly damaging effects of the use of machinery.
- Areas where *Molinia* is present at more than 20-30% cover are best not burnt, because this will encourage this grass.

In England, peatlands are offered protection under Defra's voluntary Heather and Grass Burning Code (Defra, 2007), which describes the minimum standards for environmental good practice in burning. This advises land managers against burning in sensitive areas, including blanket peat bogs and wet heathland, within 5 m of watercourses, including grips, and in other areas where there is considered to be a high risk of soil erosion. In Scotland, the Muirburn Code (SEERAD, 2001a) also recommends that burning should not be carried out on blanket bogs unless heather constitutes more than 75% of the vegetation cover. The supplement to the Muirburn Code (SEERAD, 2001b) goes on to explain that "in the latter situation, the peat is likely to be relatively dry and bog moss (*Sphagnum*) cover is likely to be sparse. Conditions that permit good control of fires are exacting and infrequent on peat ground: either much material is left unburnt, and heather regeneration is poor, or the effects are too intense and the underlying peat is exposed. In very dry [and/or windy] conditions the fire may burn uncontrollably and lead to the ignition of the peat. Once the peat ignites it may burn for months and is virtually impossible to put out [in a controlled way]. A fire that burns into the peat will cause considerable damage, which will be long lasting [consuming considerable soil carbon stocks] and could lead to serious peat erosion."

Currently, in England, blanket bogs and wet heathland should not be burned other than in line with a management plan agreed with Natural England. Such management plans being likely to involve careful burning on long rotations, using cool burns leaving large amounts of 'stick' and not damaging the moss layer (Defra, 2007). If burning must be undertaken, then it should be no more than once every 20 years, and it should be carried out when the litter/*Sphagnum* layer is wet (Tucker, 2003) or frozen. The Defra code also states there should be a strong presumption against burning on slopes greater than 1 in 3 on blanket mire or wet heath, in areas of late mature/degenerate heather, when heather cover is less than 50% and adjacent to watercourses including grips. However, there is hardly any experimental evidence on management impacts in relation to slope. In particular, more research is needed to investigate direct and indirect (e.g. via PFT and vegetation cover changes) effects on DOC and POC through erosion and runoff. The code is applied to all moorland managed under agri-environment agreements, but individual agreements may be more stringent to protect the peatland interest (Natural England, 2010). Also, moorland burning is subject to Part 1 of the Wildlife and Countryside Act which states that "it is unlawful to disturb or destroy wild birds or other protected animals, plants and habitats" (Defra, 2007).

Within the UK, there has been considerable debate on the potential effects of different land management practices on peatland C stores, with grouse moor management (and particularly muirburn) sometimes being a focus of such debate (Pearce, 2006; Yallop et al., 2009). Apart from fires consuming peat itself (out of control 'hot' fires), it is often reported that even well-managed burning can have a negative impact on the carbon dynamics of blanket bog. For example, Garnett et al. (2000) showed there was significantly less peat accumulation under a 10 year burn regime compared with unburnt plots. Ward et al. (2007) found increases in gross ecosystem CO_2 fluxes of both respiration and photosynthesis in burnt treatment plots and reduced C stocks above ground and in surface peat, relative to controls. On the other hand, studies by Farage et al. (2009) suggest that well-managed burning on upland heather moorland may not have the major detrimental effect on the carbon budget that is often assumed. However, Legg et al. (2010) questioned these results, particularly the amount of remaining biomass [C] after a fire and estimated erosion losses in relation to moisture and thus bulk density changes. Results from the only other studies addressing the impact of management on the C budget by Garnett et al. (2000) and Clay et al. (2010b) provide opposing results as to C sink or source status of burnt blanket bog. However, the latter study lacked replication and did not measure all components (i.e. POC and CH_4) or had to scale up fluxes based on less robust approaches (net CO_2 chamber flux). Clearly however, removal of biomass through burning reduces the input of litter (and thus carbon) into the soil organic matter pool. Other mechanisms, such as exposed peat surface leading to increased runoff and erosion and thus export of surface peat, are also potential factors resulting in reduced C accumulation (especially likely on slopes). Moreover, charcoal (assumed to be inert to decomposition) additions require further research as they might lead to C long-term accumulation (see Clay et al., 2010b). Particularly together with ash, charcoal could also affect physical (e.g. bulk density), chemical (e.g. reaction sites) and biological (e.g. microbial activity) properties of peat, and thus C storage and decomposition, reducing any carbon losses from burnt biomass.

The inclusion of fluvial fluxes of DOC and POC is critical to assessing the overall C balance of peatlands, and extensive correlative studies at the catchment scale across Yorkshire during 1995-2006 indicate that muirburn, in areas with *Calluna* vegetation cover on deep peat, results in marked DOC increases and water colour decline (Grayson et al., 2012; Grant et al., 2012). Holden et al. (2012) provide a more detailed synthesis of data on DOC. Notably, Worrall et al. (2010b) considered that there was a disconnection between plot-level and catchment-scale studies on burning effects on DOC, requiring further linked-up research. A number of studies also point to the importance of effects of vegetation type on GHG balances or other peatland ecosystem services**.** For example, *Sphagnum* dominated vegetation with a high water table has been shown to have GHG benefits over heather dominated deep peat (Lindsay, 2010; Couwenberg et al., 2011), whereas sedge domination tends to lead to high CH_4 emissions. Thus if management alters the vegetation cover, it is likely to alter the GHG balance (Worrall et al., 2010a). However, more work on such PFT indicators is needed.

Based on a simplified version of the Durham Carbon Model (described by Worrall et al., 2011), Natural England (2010) estimate that rotational burning of blanket bogs releases 2.56 tonnes CO_2-e ha^{-1} yr^{-1} (a total of 0.26 Mt CO_2-e yr^{-1} for England as a whole) compared with a net uptake of 4.11 tonnes CO_2-e ha^{-1} yr^{-1} for undamaged blanket bogs. However, it should be noted that, unlike the MILLENNIA model (Heinemeyer et al., 2010), the Durham Carbon Model (like most current soil decomposition models such as RothC, Century or ECOSSE) does not consider topography-linked erosion nor does it consider the total (millennia-old accumulated) peat depth in relation to a dynamic water table on a cohort basis (Clark et al., 2010). Clearly such cohort models are needed to overcome current model limitations (Dungait et al., 2012), and more data on C fluxes (i.e. DOC and POC) in relation to vegetation and topography is needed to better parameterise the models. In a recent review by Grant et al. (2012), it was concluded that insufficient evidence is available to allow conclusions to be reached on whether rotational muirburn causes major detrimental effects on overall C stores, after accounting for suggested, but unquantified, benefits that may accrue from reduced risks of wildfires. However, there is growing evidence that intensive rotational muirburn (involving short rotations) on deep peat causes marked increases in DOC (and likely also POC) export (see below).

2.2.1. Erosion

Research is lacking into the role of topography, slope and erosion, and their interaction with burn management and peatland carbon and water dynamics in general. Defra's Heather and Grass Burning Code states there should be a strong presumption against burning sensitive areas such as steep hills and gullies where the slope is greater than 1 in 2 (or 1 in 3 on blanket mire or wet heath) (Defra, 2007). Lindsay (2010) notes that studies frequently indicate 'slope' in combination with other mechanisms as a possible cause of erosion, and Carling (1986) calculates that, although peat should theoretically be stable on slopes up to 25°, slopes greater than 7° are frequently unstable, especially downslope of peat pipes. This slope-dependent reduction in total peat depth and C stocks was also be shown by comparison of model predictions to site data at Moor House NNR (Heinemeyer et al., 2010), and has also been noted by the Peatscapes project as one of the key factors explaining national peat C stock variability (pers. communication A. Hanlon; Yorkshire Peat Partnership, York). Di Folco and Kirkpatrick (2011) measured carbon losses over 4 years after a low severity planned burn, and 18 years after wildfire, on Australian peat sites, showing that planned fire increased C losses and DOC in streams. This effect varied significantly with dryness and topographic wetness index (which were used to quantify topographic control on hydrological processes). They note continued losses and slow recovery after fire on slopes compared to shelves and basins. Saturation surface excess flow has been shown to dominate gentle slopes, with near-surface rather than surface flow increasing on steeper slopes (Holden and Burt, 2003). Holden (2009a) suggests that topographic controls on peat structure in top and toe slopes promote more pool systems and more variable bulk density (BD) and hydraulic conductivity (K_s) compared to steeper mid slopes, suggesting a mechanism for different stability, preferential flow paths and ultimately gully formation, through erosion and POC export.

Fluvial peat erosion is an important issue for water supply in the UK, where DOC has increased substantially over recent decades. Yallop and Clutterbuck (2009) and Clutterbuck and Yallop (2010) found a significant correlation between the extent of managed burning and DOC export, comparing stream water data and historic aerial imagery of the Pennines over four decades. They note that declining sulphur deposition and temperature increases, as previously hypothesised (Evans et al. 2005) were less significant. Chapman et al*.* (2010) agree with this in their analyses at the Nidd catchment, but they note no correlation between the proportion of burn and *increase* in DOC and they suggest that catchments with greater connectivity between organic and mineral horizons show the greatest increase in DOC export (1986-2006). Although bare or eroded peat sites can be associated with greater runoff and erosion (e.g. Evans et al., 1999), there is not much detailed knowledge about actual rates of C export as DOC or POC in relation to different management. In particular, in the UK there are so far no published studies on POC fluxes from prescribed fire areas (Worrall et al., 2010b). In a study of water discolouration in the headwaters of the Derwent catchment, south Pennines, O'Brien et al. (2006) found there was no significant relationship between the percentage area of prescribed burn and true (filtered) water colour. Holden et al. (2012) conclude that differences between studies in burning impacts on DOC production and water colour may reflect their short time scales, failure to include shallow depths, and hydrological interconnectedness. Therefore, more work is needed on management impacts on both DOC and POC, particularly considering topography and catchment scales.

2.3. Cutting of heather

In circumstances where burning may not be an appropriate management practice, cutting of heather by mechanical means is another acceptable way of managing heather regeneration (DARDNI, 2011; Ward et al, 1995). Cutting is mostly used to encourage heather regeneration and covers a range of mechanical techniques including cutting, swiping and flailing, with machines which use either blades or chains. Swipes and flails are more suitable for woody heather and on rough ground, either using rapidly rotating chains or blades, sweeping horizontally in swipes or vertically in flails (MacDonald, 1996). Flails produce a cleaner cut than swipes and, if used at high rotation rates, produce finely fragmented brash. Forage harvesters also give a clean cut and the brash can be collected and carted away in one pass or blown over the cut area. However, it may be difficult to manoeuvre forage harvesters, particularly on uneven or sloping ground, and they are likely to suffer more from damage than flails or swipes (MacDonald, 1996). In any case, if brash is to be left on site, it needs to be chopped very fine and evenly distributed to avoid smothering regrowth and other vegetation. A potential advantage of such cutting practice is the reduced actual evapotranspiration (AET) from the covered ground compared to exposed burn areas, also limiting any negative rain/erosion effects. This is the premise underlying the use of brashing for restoration purposes.

Cutting might be expected to produce faster initial rates of vegetative heather growth than burning (Liepert et al., 1993), since a greater number of sprouting centres are retained (Mohamed and Gimingham, 1970) and resprouting heather plants grow much more quickly than heather seedlings (MacDonald, 1996). However, resprouting from dormant buds on the stem base declines as bushes become larger and more woody (Hobbs & Gimingham, 1984), so that cutting is more likely to be suited to vigorous building or early-mature heather stands (MacDonald, 1996) if future heather cover is to be high. As reported by Liepert et al. (1993) and Cotton and Hale (1994), in old stands of heather, cut areas may take longer to regenerate heather cover than burnt areas, and sometimes may not regenerate at all (Backshall et al., 2001). This is because in old stands of heather, regeneration is almost entirely from seedlings whose growth is slower than that from stem regrowth (Liepert et al., 1993) and hence regeneration is worse in cut plots than burnt plots (Liepert et al., 1993; MacDonald, 1996). The pulse of high temperature experienced during burning is thought to stimulate seed germination and vegetative growth of heather and a number of other species (Whittaker and Gimingham, 1962), an effect which is lacking when cutting is used alone. Also, if the brash is left behind after cutting, this can inhibit seed germination and establishment due to smothering (MacDonald, 1996). For these reasons, MacDonald (1996) argues that cutting is less suitable than burning for restructuring and regenerating old heather stands, a claim supported by Liepert et al. (1993), which compared heather regeneration after cutting (with brash removal) and burning on old *versus* young *Calluna* stands in the North York Moors. However, this was in marked contrast to a previous assertion of the Moorland Association[1] that mature, and especially degenerate, heather will regenerate more quickly after cutting than after being burnt. There is only one study so far comparing burning *versus* cutting effects with brash being either removed or left on site (Worrall et al., 2013); this study in the Peak District revealed lower water tables under burn than under mown management and lowest under intact vegetation (reflecting differences in evaporation losses). However, the evidence on heather regeneration after cutting compared to burning in truly replicated studies is poor and clearly more research is needed.

A clear advantage of cutting is that this activity is much less constrained by weather conditions than burning (Tucker, 2003), although access for cutting machinery will be easier when conditions are drier and the ground is firmer (MacDonald, 1996). Areas of moorland next to forestry or other fire risk habitats can be cut to minimise the risk of a fire spreading, as can areas adjacent to roads to avoid the risk of smoke becoming a traffic hazard (MacDonald, 1996). Although cutting is generally more expensive than burning, the sale of cut heather for commercial purposes can reduce the overall cost (Backshall et al., 2001), particularly if the heather is reasonably convenient to where a lorry can be loaded (Moorland Association[1]). However, not all areas are accessible to large scale mowing or even less so bailing equipment. The capital costs of cutting are high and operational costs per unit area, using a contractor, are likely to be about 20% higher than burning with cut firebreaks and foam traces, which in turn are about 70% higher than traditional controlled burning (MacDonald, 1996). However, if labour and suitable machinery are already available, cutting or swiping can be done for about half the cost of traditional controlled burning (Tucker, 2003). Furthermore, costs to society and the environment through 'out of control' fires are difficult to measure and are never included in such calculations. Cutting has become increasingly important as a substitute for burning, as well as being used as an additional tool to assist moorland burning practices (Backshall et al., 2001), for example:

- in the cutting of fire breaks;
- in breaking up large areas of old, leggy heather and allowing the re-establishment of a burning programme;
- in providing a practical alternative to burning where this is not possible or desirable;
- in cutting of drain sides to maintain visibility for safety;
- in fitting into the other demands of the farm when convenient and when labour is least in demand elsewhere, e.g. in January or February; and
- in helping to control purple moor-grass where burning would be unsafe.

Cutting may also be an effective alternative to burning in vegetation containing a high proportion of purple moor-grass, where removal of the litter and grazing the new grass growth can lead to increased species-richness (Backshall et al., 2001). No other research was found reporting impacts that may result from the interaction of grazing and cutting. It is important to note that currently cutting is often seen mainly as a tool for achieving optimum heather re-growth and coverage alongside, for example, fire breaks. It could, however, also be used to 'regulate' heather re-growth and encourage 'active' bog vegetation amongst heather by changing the frequency of mowing, and removal or non-removal of the brash, and it may provide benefits for water colour.

There are some serious drawbacks to cutting. There are many areas where the ground is too steep, rocky, or wet for machinery to be used safely or without causing damage to the vegetation and peat. In drier parts of the country, the cut material may not decay quickly, suppressing regeneration of the heather if a thick mat of cut material is produced. Cutting machinery also can cause damage to some archaeological features unless used with great care. Cutting, like burning, is a drastic event for the vegetation and its associated fauna. The machinery used in cutting can damage fragile peaty ground and is restricted to areas with suitable topography (e.g. accessible, not too steep, uneven or too rocky). In practice therefore, cutting is mainly suited to level, dry heath conditions and

[1] http://www.moorlandassociation.org/heather_burning2.asp (accessed 2012 but no longer available)

other disadvantages (mainly from a grouse management viewpoint) of cutting according to the Moorland Association[1] include:

- Any access with cutting machinery runs the risk of causing compaction damage not just on the area of the cut but in accessing that area. If the cut material is baled and removed this problem can lead to rutting and soil damage.
- Very thick areas of cut material, if left as a mulch, can provide too thick a mat layer on the ground that hinders regeneration.
- Although regeneration from root stock, particularly if the heather is degenerate, is quicker after cutting, the regeneration from seeds is enhanced by fire and therefore is much slower to occur and overall seed germination is less. A greater regeneration from grasses rather than heather may therefore occur.
- Some people feel uneasy about the sight of large, regular cut shapes across the landscape, particularly on exposed visible hillsides, which look unnatural and perhaps jar the eye more than the edges of burns.
- When the cut material is removed it also removes nutrients from the land.
- Higher carbon footprint (due to burning of diesel and a high wear rate on the cutting machinery and tractor).
- Staff need to be sufficiently trained in the safe use of the cutting machinery and in how to carry out cuts in accordance with the management plans.
- Any access with machinery runs the risk of damaging buried archaeology, both that which is known about and that which is yet to be discovered.

The Moorland Association (MA) concluded that, depending on conditions of the ground and available resources, a mix of both cutting and burning is likely to be required in order to deliver the optimum integrated heather / grouse moorland management.

Thus, from the MA's viewpoint, cutting is probably best regarded as a complementary method to burning, being used when and where burning is not possible or desirable, and for creating fire breaks to improve fire control (see also Ward, MacDonald and Matthew, 1995). Backshall et al. (2001) list various recommendations concerning cutting of heather relating to:

- timing (e.g. heather regeneration is best after spring cutting but must avoid nesting season; avoid cutting when ground is saturated as this may damage peat surface),
- areas to avoid (e.g. wet areas and bogs; steep and rocky ground; large areas of old heather where regeneration is likely to be poor; where archaeological remains might be damaged,
- method of cutting (size and shape of cuts; 10 cm heather stem to be left; use 4-wheel drive tractor and, on softer ground, fit double wheels to avoid compaction and peat damage; Turner flail extremely effective),
- uses of heather (either remove cut material e.g. for restoration work if cut after flowering season or use a double-chop forage harvester so that fine material will be incorporated into the soil quite rapidly).

According to the supplement to the Scottish Government's Muirburn Code (SEERAD, 2001b) the great advantage of cutting is that it is much less hampered by the weather and there is no fire risk to neighbours' property. Regeneration can be very good, although less nutritious for livestock or grouse than regrowth in the first few years after a fire. Usually, the most practical technique is to use a chain swipe mounted on a four-wheel drive 80 HP (60 kW) or 100 HP (75 kW) tractor, which can be fitted with double wheels for softer ground. This produces finely mashed-up material which appears to decay quite rapidly, especially in the wetter, western parts of the country where an alternative to muirburn is most needed. It is particularly valuable where there is much purple moor-grass among the heather. On ground suitable for the use of a double-chop forage harvester, the cuttings can be blown outside the cut patch. Alternatively, cut material can be baled and sold for a variety of mulch and filtration uses. Suitable machines include specifically designed heather flails and self-powered flails which can be towed behind an all-terrain vehicle. Alternatively, it is possible to use adapted single and double-chop forage harvesters. The flail should be set 12.5 cm to 15 cm above the ground (5 to 6 inches). As with burning, it is important to leave the side of the block as irregular as possible to result in a more natural appearance in the landscape. However, cutting is not recommended for blanket bog where regeneration is very slow and machinery can damage the vegetation (DARDNI, 2011). Also, cutting should not be carried out within 10 m of any watercourse to minimise pollution of the water and future excessive trampling of watercourse edges (DARDNI, 2011).

There has been little research carried out on the effects of cutting (on species diversity, water quality, GHG fluxes, carbon balance etc.), compared with that on the impacts of heather burning. In terms of frequency, Backshall et al. (2001) recommend cutting every 10-20 years for heather growing alone, or in mixtures with grass, but more often for fire breaks alongside forestry. No research results could be found for the impacts of repeated cutting at high frequency, or interactions of cutting with grazing and the resulting effects on heather coverage, PFTs and blanket bog species biodiversity. However, an initial repeated cut (after ~5-10 years) could be expected to give an advantage to more 'active' bog species by preventing a too rapid reestablishment of heather dominance.

The mulch produced by cutting can be removed from the site either by blowing it over the surrounding area if using a forage harvester, or by raking it into piles and drawing it off the site. A layer of mulch may suppress heather regeneration as well as limiting evaporation losses (thus keeping sites wetter), depending on the chop length, the quantity of material left behind, the timing of the cutting and the amount of rainfall and weather in the area (DARDNI, 2011). Nonetheless, brash could smother slower growing species and prevent seed germination so should not be too thick, although there are only limited experimental data available on such effects (e.g. Liepert et al., 1993).

Finally, although cutting of heather, and any subsequent bailing, are not subject to the burning season, these activities are subject to Part 1 of the Wildlife and Countryside Act which states that "it is unlawful to disturb or destroy wild birds or other protected animals, plants and habitats" (Defra, 2007). Heather cutting should not be carried out after the 15th April nor throughout the summer months, as ground-nesting birds will be present (SEERAD, 2001b). The use of cutting machinery on a Site of Special Scientific Interest may also be an offence if the use of vehicles has been identified as a 'potentially damaging operation' and consent for their use has not been given. Other legal obligations relating to the safe use of machinery will also apply.

2.4. Herbicide

In theory, herbicides such as Asulam could be used to control heather in the same way as formerly used to control bracken or *Molinia*. However, there are numerous environmental concerns about the use of herbicides to control moorland vegetation. Spraying-related water quality issues (Holden et al., 2008b) and ecological implications are of immediate concern, as in the UK, especially in England, most blanket bog areas are within major drinking water supply areas and also part of ecologically important areas for biodiversity. Use of the herbicide Asulam (marketed in the UK as Asulox) has recently been banned by the European Commission, which emphasises the growing concerns about using herbicides for such vegetation control (although concern is mostly focussed on agricultural crop applications), and Asulam is specifically identified as a high risk to birds. Consequently, although Asulam might be re-approved by the EU but unlikely before 2017 (see previously available at: http://www.cla.org.uk/rpro/Asulambriefingnote.pdf; accessed November 2012; similar document at: www.environmentdata.org/download/file/ealit:1941/ealit:1941/1/0/OBJ/PDF/NA), the environmental concerns (i.e. water quality) in ecologically sensitive areas (i.e. SSSIs), as well as potential cross- contamination between experimental plots during and after applications, seem to prohibit further consideration of such control measures.

2.5. Other potential management options

Notwithstanding the interesting prospect of knowing what the result of a ‘do nothing’ option would be (Annex A.1), there are a few other potential control measures, which warrant further consideration:

a) **Natural bio-control**: naturally occurring heather beetle (*Lochmaea suturalis*) damage, or even its targeted introduction could be considered, as was done in New Zealand (Peterson et al., 2004). However, its application would require development of some level of control to prevent ‘out of control’ effects beyond target areas. Whilst Berdowski (1987) observed the initiation of a change from heather to grass dominated vegetation by outbreaks of infestation with heather beetle, a study in Scotland showed fast recovery of heather after a beetle infestation, although this was suppressed where a moss layer was present (MacGillivray, 2004). Therefore, acceptance of naturally occurring infestations with no subsequent restoration efforts on bog or wet heath should probably be encouraged.

b) **Repeated pressure on vegetation cover**: this is based on field observations that heather does suffer from repeated crushing along vehicle tracks (even from low PSI vehicles) or foot paths (even from irregular monitoring). However, no literature on applying some form of mechanical crushing as part of management could be found.

c) **Re-introduction of *Sphagnum* species**: the effectiveness of this has been shown to relate linearly to coverage of bare ground (Rochefort, 2000), which might also apply after disturbances such as burning or mowing, when peat surfaces can be exposed. Robroeka et al. (2009), and more, recently the Yorkshire Peat Partnership (A. Hanlon, pers. comm., Yorkshire Peat Partnership, York), showed the potential to reintroduce key *Sphagnum* species to create ‘active’ bog communities. Importantly, *Sphagna* are reported to function as indicator species (e.g. of pH, moisture and nutrient levels) for blanket bog functioning (O’Reilly, 2008). However, various methods (e.g. gel pellets, plugs) of re-introduction are currently being tested.

d) **Liming**: to encourage vegetation (re-)growth (Anderson et al., 1997) or increase catchment pH (Jenkins et al., 1991) lime has been applied, but subsequent negative effects on *Sphagnum* (Clymo, 1973) and possibly other bryophytes can be expected. Liming could also stimulate decomposition and thus lead to a decreased C balance and increased DOC levels (Tipping et al., 1989).

e) **Combined burn and grazing management**: Although interactions have not been quantified apart from one study at Moor House (Garnett et al., 2000), particularly on blanket bog (see Shaw et al., 1996), it is clear that sufficiently high grazing levels on recently burnt areas will cause a shift from *Calluna* to grass dominance (Thompson et al., 1995). However, high level grazing on burnt areas might lead to substantial erosion, whereas low intensity grazing has been reported to reduce heather coverage (Rawes and Williams, 1973, cit. English Nature, 1996).

3. Possible effects on ecosystem services

Based on the review of the literature presented in Section 2, there seem to be several key effects on ecosystem services which are of specific relevance to this project:

a. Erosion and water quality

Analyses of hydrograph records from the 1950s to the present show higher mean values and higher peaks per unit of rainfall from a UK peatland catchment, which coincide with increasing bare peat (Grayson et al., 2010). The scale of analysis in relation to management impacts on runoff and flood risk is reviewed by Pattison and Lane (2011), who note that the impacts of localised management affecting hydrological connectivity may not be detected at the greater catchment scale, where climatic influences become more dominant. Holden and Burt (2002) investigated how plot-scale surface cover affects run off and sediment production with a 1 m^2 rain simulator at different intensities, first in spring, and subsequently after a warm dry period. They found a significant difference in runoff from *Eriophorum*, *Calluna* and bare plots during the dry period, but not from *Sphagnum*, which they suggested protects the surface from desiccation. However, the authors did not measure effects on DOC and POC.

Gullies and peat pipes are an important source of fluvial erosion. Holden (2009a) found that macropore (>1 mm) flow contributions, as a proportion of saturated hydraulic conductivity (K_s), were more important over peat than less organic soils, with spatial patterns of macropore flow reflecting slope position. The importance of management around slopes and gullies is shown by Holden et al. (2007), who found natural infilling of drains occurred on gentle (<4°) slopes, but rarely on steeper (>4°) slopes, and rarely when incised into the mineral substrate, and by Worrall et al (2011), who showed the C budget benefits of re-vegetation following wildfire, particularly for reducing fluvial POC losses. Although there are estimates of wind erosion rates from exposed areas, hardly any data are available on the actual amounts of erosion due to runoff as DOC and POC, particularly considering the effect of burning and subsequent exposure of peat to rainfall.

Clay et al. (2012) showed elevated water colour, but no change in DOC levels, in years following burning, in plot-scale soil pore and runoff water over varied burn ages, and proposed that burning affects composition of the DOC. Worrall et al. (2013) found reduced DOC concentrations in peat water after burn and mow, although they did not have pre-management change measurements. However, the differences could be explained by covariates; the DOC concentration of soil water samples rose with increasing depth to the water table and with increasing soil water conductivity. However, Worrall et al. (2013) found the E4/E6 ratio (water quality measure based on UV absorption spectra) to be lower on burnt sites, implying that higher molecular weight and more humified DOC was present; crucially, management (both, burn and mow), but not slope, affected surface runoff. Impacts on biodiversity include POC-induced reduction in gilled invertebrate numbers after burning (Ramchunder et al., 2009; 2012).

A key question is how vegetation cover or plant species affect erosion, and its interaction with management. Holden et al. (2008a) compared overland hillslope flow velocity along 6 m runs between four PFTs; they showed the velocity over *Sphagnum* to be significantly lower. This highlights the potential for PFT management effects on ecosystem services around carbon stocks and water quality. Vegetation community change in response to burning can be brought about in the long term, by heather dominance leading to drying as a result of increased evapotranspiration, and lowering of the water table, thus impeding *Sphagnum* growth (Worrall, et al., 2007, Lindsay, 2010). Heather removal has been shown to raise the water table (Worrall et al., 2007; Worrall et al., 2013). Holden (2005) showed, under controlled conditions of topographic index, that peat depth, water table depth and frequency of piping was greater under the *Calluna*-covered peat and bare peat than under other vegetation covers. However, it is unlikely that current PFTs accurately reflect the eventual long-term (millennia-scale) natural vegetation cover; particularly, heather dominance might only reflect the last two hundred years – a direct result of burn management and livestock grazing (pers. comm. Astrid Hanlon, Yorkshire Peat Partnership).

Eriophorum and other sedge species are recognised as early natural colonisers of bare peat, and creators of a suitable microclimate (increased humidity, shade, stability) for the subsequent establishment of diverse bog species, particularly *Sphagnum* (Rydin and Jeglum, 2006). Armstrong et al (2012) found different DOC concentrations in plot pore water associated with *Calluna* (high), *Sphagnum* (low) and sedge (medium). They further note that *Calluna*-dominated drains had greater DOC concentrations than pore water, whereas in sedge-dominated areas it was greater from pore water than in drains. There is now some evidence to link *Calluna* dominated vegetation to DOC export and, therefore, the potential to manage plant functional types (PFTs) to reduce rising DOC in stream waters. However, the mechanisms and processes of this remain uncertain.

b. GHG balance and methane emissions

Methane emissions are an important component of the carbon budget of natural and restored blanket peatlands (Baird et al., 2009). Some vascular plants (with aerenchyma tissue) growing in bog pools and hollows can act as a direct route for methane (CH_4) release, thus bypassing the oxidising layer in the acrotelm, whereas *Sphagnum*-dominated swards can suppress methane efflux through oxidation (Lindsay, 2010). There are very few full C or GHG budget studies for peat soils in the UK, with a particular lack of data for both CH_4 and N_2O fluxes (Worrall et al., 2011). There are also very few studies which take explicit account of the relationship between methane emission levels, vegetation and associated topography (i.e. slope, aspect) or micro-topographical structures (e.g. pools, hollows, hummocks). Nonetheless, any management interventions on blanket bog that change vegetation composition, water table depth or micro-topography may have implications for net methane fluxes and should be investigated further.

However, it is noteworthy that most previous GHG emissions data are likely to be flawed, thus leading to false estimates. For example, Fig. 3.1 in Baird et al. (2009) highlights three potentially serious issues with the commonly used static chamber GHG measurements: (1) clear chambers leading to overheating during long closure times and thus increased fluxes through Q_{10}-related temperature sensitivities of respiration; (2) deep collar insertion cutting roots and thus increasing material available to decomposition whilst reducing fresh root-derived C inputs; (3) collar insertion leading to interference with water flow, effectively preventing lateral drainage and trapping water and likely animals in the collar area (for a full review of collar depths and an experimental study on these impacts see Heinemeyer et al. (2011)).

c. Biodiversity

Burning and grazing impacts on plant and animal diversity are the most studied and a review is provided in a report by English Nature (1996). Overall, effects depend on the state of the peatland (wetness) and the intensity of the burn and grazing regime and their interaction is complex and not well studied. However, rare bird species seem to benefit from the current mosaic burn management.

4. Recommendations for possible treatments

Recommendations for experimental treatments were developed which would most effectively address the knowledge gaps and policy priorities identified in the literature review. These were discussed in detail at a stakeholder workshop (reported in **Annex A.1**) and then modified. The final recommendations are summarised below.

The most likely beneficial impact on supporting 'active' blanket bog vegetation is to be gained from grip blocking. For any experimental comparison within this project, a similar density of grips (ideally blocked) is therefore important. As large areas of the UK blanket bogs are under burn rotation for grouse management, which have recently or will soon be grip blocked, it is clearly important to include such areas in this project. Moreover, as most sites contain some *Sphagnum* moss, yet might be missing key peat forming species, the addition of specific *Sphagnum* re-introductions was recommended at the plot-scale. Initially, this review recommended considering an exclusion of grazing treatment. However, this plot-level treatment was felt to be too vague in this context (considering the complex nature of grazing impacts, see Section 2.1) and was therefore rejected during the stakeholder workshop (Annex A.1). Instead, grazing exclusion plots were dropped in favour of a 'do nothing' uncut control plot to allow for a more balanced experimental design. Based on the above information and the project workshop, it was concluded that in addition to comparing a 'business as usual' burn rotation (with plot level monitoring) to alternative mowing with leaving brash at the catchment scale (to capture landscape scale impacts on hydrology, water budgets and water quality) the following five treatments should be investigated at the plot-scale (5 x 5 m) within the mown catchment (to allow investigating process level processes and vegetation specific impacts):

1. **Mowing with brash left on site (T1)**:

   Cutting is now generally an accepted measure for heather dominated blanket bog and its careful application should encourage 'active' blanket bog forming species whilst avoiding negative burn effects. A fine brash layer could suppress quick regeneration of heather whilst also holding back water.

2. **Mowing with brash removal from site (T2)**:

   This would offer testing against T1 for the effect of smothering plant and seedling growth by the brash layer and its effects on evaporation losses between different plant community ground cover.

3. **(Mowing frequency with) brash left on site and added *Sphagnum* (T3)**:

   Repeated cutting (either shorter or possibly longer intervals than in T1+T2 depending on project length) could encourage 'active' bog species by preventing re-establishment of heather dominance. In addition it would test if additional encouragement of *Sphagnum* species is possible, firstly in comparison to T1+ T2, and, secondly at different mowing frequencies.

4. **(Mowing frequency with) brash removed from site and added *Sphagnum* (T4):**

   This enables trialling *Sphagnum* introduction as part of a mowing regime, without a dense brash layer compared to T1 + T3 – mowing frequency depending on project length.

5. **'Do nothing' uncut comparison (T5; formerly exclusion of grazing treatment)**:

   There is a clear lack of information on the 'do nothing' uncut scenario and any treatment would benefit from a comparison to what would have happened anyhow – this would specifically include climatic events during the monitoring after the treatment application phase (e.g. natural drought or wet years), and how different benefits are in respect to growth and ecosystem services. Moreover, it would offer a crucial additional 'control' to the burn and other treatment areas and is mainly to be seen as a necessity for a balanced experimental approach, also allowing valuable comparison of, and modelling information on, for example, carbon balance, growth rates, regeneration and nutritional value of heather.

Notably, an additional control (burn) plot would ideally also receive the same *Sphagnum* treatment as the mown plots (i.e. T3 and T4 above) to compare *Sphagnum* pellet establishment on a burnt *vs*. mown surface.

The prepared experimental platforms at three sites in Northern England offered a rigorous experimental and replicated test-bed at both the catchment and plot-scale for the identified management options. However, as is the case with any field experimental work, due to the ecological complexity, the expected impacts of any of the above options will inevitably be also influenced by other environmental factors beyond the project's control such as climate (e.g. drought) or pests (e.g. heather beetle). Moreover, preventing inter-treatment influences (e.g. edge effects) needed to be addressed when considering treatments and their applications; therefore, a 5 m buffer zone was set up between the individual treatment plots.

Crucially, the experimental approach proposed follows O'Brien et al.'s (2007) recommendation for a long-term experimental monitoring commitment aimed at understanding the impacts on blanket bog and the time frames needed before blanket bogs begin to recover and return to favourable, 'active' conditions. Clearly, timescales are important and although some changes can be expected within the initial 5-year project period, a continuation beyond five years is likely to be needed to assess transient effects *versus* more permanent shifts in plant community responses and associated long-term impacts on ecosystem services. To our knowledge, BD5104 offers the only such current UK long-term replicated plot and catchment-scale experiment (on blanket bog) to provide such a test platform and as such is an exciting opportunity for all stakeholders involved.

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