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Environmental change in moorland landscapes

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Abstract

Moorlands are unique environments found in uplands of the temperate zone including in the UK, New Zealand and Ireland, and in some high altitude tropical zones such as the Andean páramos. Many have been managed through grazing, burning or drainage practices. However, there are a number of other environmental and social factors that are likely to drive changes in management practice over the next few decades. Some moorlands have been severely degraded and in some countries conservation and restoration schemes are being attempted, particularly to revegetate bare soils. Native or non-native woodland planting may increase in some moorland environments while atmospheric deposition of many pollutants may also vary. Moorland environments are very sensitive to changes in management, climate or pollution. This paper reviews how environmental management change, such as changes in grazing or burning practices, may impact upon moorland processes based on existing scientific understanding. It also reviews the impacts of changes in climate and atmospheric deposition chemistry. The paper focuses on the UK moorlands as a case study of moorland landscapes that are in different states of degradation. Future research that is required to improve our understanding of moorland processes is outlined. The paper shows that there is a need for more holistic and spatial approaches to understanding moorland processes and management. There is also a need to develop approaches that combine understanding of interlinked social and natural processes.

Keywords: uplands; moorland ecology; hydrology; water quality; afforestation; burning; peatlands; bogs; heath; heather; grazing

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Contents

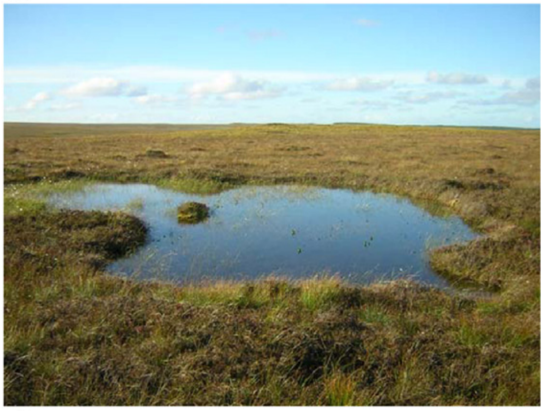
1. Introduction

Moorlands are unique environments found in uplands of the temperate zone including in the British Isles, New Zealand, Japan, Scandanavia, Russia and Tasmania and in some high altitude tropical zones such as the Andean páramos (e.g. Buytaert et al., 2006). Moorlands are open areas with acid or strongly base-deficient soils such as peat. Their hydrology, soils and ecology are very sensitive to small changes in the local environment. In many areas the management of moorlands has led to severe degradation with erosion, flooding, poor water quality and loss of ecological biodiversity. It is therefore important to understand how moorland environments may respond to future management strategies or changes in climate or atmospheric pollution. This paper aims to review our current understanding of moorland science and uses as its focus the UK uplands as a case study site where there are moorlands in different states of degradation.

Moorland covers around 38% of Scotland, 5.5% of England and Wales and 8% of Northern Ireland. Here the moorland habitats are varied but can be categorised as: (1) heathland communities characterised by small shrubs such as ling heather (*Calluna vulgaris*) and bilberry (*Vaccinium myrtillus*) (Gimingham, 1972); (2) mire or bog communities characterised by mosses (dominated by *Sphagnum* spp.), sedges such as cotton-grass (*Eriophorum spp*.) and small shrubs (Johnson and Dunham, 1963; Rawes and Heal, 1978); or (3) acid grassland communities characterised by grasses such as *Deschampsia flexuosa*, *Festuca ovina* and *Nardus stricta* (Pearsall, 1965). Often moorlands are known for their dominant vegetation (e.g. 'heather moors', 'bilberry moors' or 'sedge moors'; (Pearsall, 1965)), or the animals for which they are managed (e.g. 'grouse moor'). Variation in UK moorland plant communities results from: (1) north–south and altitudinal climate variation; (2) east to west precipitation gradients (heaths towards the east and bogs towards the west and north); (3) local drainage conditions; (4) prescribed fire management; (5) wildfires; (6) grazing pressure; (7) other management (afforestation, drainage, peat cutting etc), and (8) acidic deposition. Fig. 1 shows some examples of typical UK moorlands. Table 1 provides further information on typical moorland plant communities and environmental conditions. The UK climate means that its moorlands are subject to seasonality in temperature and generally have a greater than six month growing season. Precipitation tends to be greater in the winter, and is synoptically controlled. Snowfall can be common, but there is rarely a lying snowpack for more than two weeks at a time across all but the very highest moorlands.

While UK moorlands form the focus of this paper, it would be useful at this point to note that moorland environments occur in other regions to enable the UK example to be placed within a broader context. Moorlands are more prominent in the northern hemisphere mid to high latitudes. Nevertheless they are also found in upland equatorial areas and southern hemisphere temperature zones. The UK moorlands are similar in appearance to moorlands elsewhere in the world although the species and climate in each area are different. For example, moorlands known as páramo cover the upper parts of the northern Andes between 11° north and 8° south latitude where there is very limited seasonality and regular supply of rainfall all year round (Buytaert a)

b)



c)

Fig. 1. Typical moorland scenes in the UK; a) recently burnt moorland dominated by *Calluna vulgaris*; b) a *Sphagnum*-dominated blanket bog with pool complex c) a drained moorland dominated by sedges and grasses.

et al., 2006). The páramo tends to have high speciation and an exceptionally high endemism. 60% of species are endemic and adapted to the specific physio-chemical and climatic conditions, such as the low atmospheric pressure, intense ultra-violet radiation, and the drying effects of wind. These environments support grazing, coniferous afforestation and potable water supply, but are often degraded by land drainage, overgrazing and liming and there are problems of deteriorating water quality (Buytaert et al., 2005). These are the very same uses and problems that face the UK moorlands.

Many moorland areas support globally rare species and are nationally important such as golden plover, dunlin and peregrine. Seventy-five percent of the world's heather moorland is in the UK uplands (Tallis et al., 1998). UK moorlands are not only rare and important habitats, they also support multiple land uses, i.e.: (1) water collection: they are source areas for most major UK rivers and potable supply; (2) agriculture: they are used for low-density sheep and deer farming (particularly in Scotland) as well as for cattle farming in places; (3) commercial forestry: around 20% of the UK moorlands are now afforested with coniferous plantations; (4) sport and leisure: through maintenance of a suitable habitat for game shooting (such as red grouse); and (5) tourism, attracting millions of visitors to upland areas. For example, the Peak District National Park located in north-central England is proximate to large urban areas, including Manchester and Sheffield, and receives 22 million visitors per year (Peak District National Park Visitor Survey, 1998). In 1998 a tourism employment survey estimated that the overall business turnover arising from tourism in the Peak District National Park was £75 million. Within the National Park the estimate for visitor spending in 1998 was £185 million, which supports over 3400 jobs, representing 27% of total employment (Peak District National Park Visitor Survey, 1998). Over 50% of jobs are indirectly linked to tourism in the Park. Such large visitor pressures and the possibility that visitor numbers to the moorlands may increase in the future may also result in environmental change and force particular management strategies to be adopted in hotspot locations. Thus, moorlands have many uses, and a simplistic classification on the basis of a single land use (e.g., grouse moor, catchment area) should be avoided.

The UK's moorlands are, however, not 'natural' environments. During the mid-Holocene humans cleared woodland to create pasture (Simmons, 2002). Tree growth was kept at bay by grazers and by deliberate fire setting. In cooler, wetter areas woodland became replaced over hundreds to thousands of years by blanket peat (Holden and Burt, 2003c). This peat has developed over large parts of the gently rolling upland landscape and in some locations is over 8 m deep (Charman, 2002). In the wet north and west of Scotland the blanket peat has developed down to the coastline. While the peat bogs and heather moorlands of the UK are perceived as unchanging "natural" or "undisturbed" rural environments by most

Table 1
Major UK moorland types based on National Vegetation Classification (NVC)

Classification	Community (NVC)	Soil type and pH (averages for sub-communities)	Rainfall (mm)	Altitude (average for sub-communities)	Typical species
Heath	Heath (H9, H10, H12 to H22)	Acid to circumneutral soils, usually freely draining on base poor substrate. Often humic upper horizons. pH 4.4 to 6.4	Variable between communities. From as low as 800 mm for H9 to 1600–3200 mm (H20).	113 m to 814 m	*Calluna vulgaris* or *Vaccinium myrtillus* dominated
Mires	Blanket and raised mire (M17 to M20)	Peat pH 3.1 to 4.7	Between 800 mm and 2000 mm depending on community. Lowland raised mires (M18) having relatively low rainfall	206 m to 856 m	*Erica tetralix, Calluna vulgaris, Eriophorum vaginatum, Scirpus cespitosus, Sphagnum papillosum*
	Wet heath (M15)	Peats or acid/oligotrophic mineral soils. pH 4.2 to 5.6	At least 1200 mm generally more than 1600 mm	177 m to 290 m	*Scirpus cespitosus, Erica tetralix*
Calcifugous (acid) grassland	Grassland (U4 to U6)	Base poor soils often with peaty horizons. pH 4.0 to 5.9	800–1200 mm	181 m to 706 m	*Nardus stricta, Juncus squarrosus, Festuca ovina*

ramblers and tourists, the reality is that moorlands have been created and maintained by land management. This includes most of the UK's upland blanket peat moorlands which are unlike many of the moorlands in the Faroes, for example, for which there is growing evidence that these have developed naturally (Lawson et al., in review).

Many UK moorlands are severely degraded and restoration measures are being implemented. However, some scientists have argued that the reversal of moorland processes is not always possible because past management has led to irreversible changes in soil chemical and hydrological properties (at least in the short to medium term of human timescales) (Holden et al., 2004; Holden, 2005a,b). Many moorland restoration schemes aim to return moorlands to a habitat state similar to that of a few decades or a century ago. This is, of course, an arbitrary decision given the "non-natural" state of UK moorland environments. At the same time, other drivers such as climate change, increased flood risk and deterioration in water quality (Wallage et al., 2006) also command attention. In addition to the current multiple moorland uses, new ideas about managing moorlands for carbon and for water are emerging. This change may produce further conflicts in moorland management. For example, is the idea of burning moorland at odds with the utilisation of moorlands to maximise carbon uptake or to maintain clean water supplies? Wider socio-economic drivers of change will also affect future moorland management. These include rural depopulation, decreasing farmer incomes (Peak District Rural Deprivation Forum, 2004), increase in second homes, and European and national policies that are moving away from an emphasis on agricultural production towards a more holistic approach to environmental management (Lowe et al., 2002).

The future for UK moorlands is, therefore, difficult to predict, but it is important that we understand the impacts of current drivers of change in order to sustainably manage the moorland environments. The focus of this paper is, therefore, on improving our understanding of the impacts of current drivers of change on the moorland environment. This paper firstly identifies major drivers of change in moorlands and briefly summarises these drivers which are indeed major subject areas in their own right (land management and socio-economic change, acid deposition, climate change). Then the paper outlines how UK moorland managers currently prioritise responses to the drivers of change. The paper then reviews the scientific literature on the likely management responses to drivers of change in order to determine their impact on the local and downstream environment. Throughout the review we will identify areas where further scientific research is required. However, the final section of the paper highlights the key areas for future research.

2. Drivers of change

2.1. Land management and socio-economic change

The UK moorlands have been, and continue to be, subject to several drivers of change. These act at a variety of scales from national to local sources. The management of moorlands has changed over the last few centuries in response to economic drivers such as (1) the switch from summer moorland sheep grazing to hardier breeds able to over-winter in the hills, in the 18th Century; (2) the increase in management for moorland sport based primarily on red deer (*Cervus elaphus)* and red grouse (*Lagopus lagopus scoticus*), from the 19th Century; (3) government subsidies following the Second World War for the cutting of drainage ditches in moorlands (Holden et al., 2004); (4) European Common Agricultural Policy (CAP) subsidies for sheep farming that resulted in a 30% increase in sheep numbers on UK moorlands between the 1970s to 1990s.

The intensification of UK upland agriculture, largely due to subsidisation for sheep production has been widely blamed for the recent erosion of upland soils and losses of certain types of moorland habitat (Ross et al., 2003). Blanket peats, for example, have suffered severe degradation in many parts of the UK and particularly in the English Pennines. Erosion in the southern Pennines has been very severe over the past 200 years and has been studied extensively where it is deeply gullied and devoid of vegetation (e.g. Bower, 1961; Tallis, 1965; Labadz et al., 1991; Yeloff et al., 2005). Eroding peat causes reservoir infilling and severe management problems (e.g. Fig. 2) and is also associated with release of heavy metals that have been deposited from the atmosphere since the industrial revolution (Rothwell et al., 2005; Shotbolt et al., 2006).

Major changes in moorland vegetation have been observed over the 20th century. In Scotland one quarter of heather moorland has estimated to have been lost since the 1940s (Moorland Working Group, 1998) and replaced by coarse grasses such as *Molinia* and *Nardus*, sedge moors, bracken and coniferous plantations. For the Peak District and Cumbria, 36% of heather has been lost since the early part of the 20th Century (Anderson and Yalden, 1981; Felton and Marsden, 1990). While the patterns are not uniform, there has been a general trend from more productive vegetation with high species diversity to large areas dominated by less diverse and more aggressive species of lower agricultural value such as *Molinia* and *Nardus*. This is widely blamed on overstocking of sheep and deer, although other factors such as nitrogen deposition have also been implicated.

Fig. 2. Moorland gully erosion in the Peak District.

The rural uplands of the UK have significantly changed in demographic composition over the past two centuries. Currently, a depopulation trend is continuing as part of a broad-scale shift in the UK's economic structure but also because income from land management activities and agriculture, particularly in uplands, are considered inadequate in relation to expenditure. Both farming and grouse-shooting activities operate at the margins of financial viability, and are heavily reliant on agricultural subsidies (Dougill et al., 2006) and some 93% of the Peak District National Park qualifies for funding under the European Commission Directive for special assistance to Less Favoured Areas (75/268). With few opportunities for financially rewarding employment, younger, unskilled workers are increasingly choosing to leave the upland moorland regions, having also been priced out of local housing markets due to increasingly affluent commuter populations and an influx of second-home owners. The impacts have been both social and environmental, as the remaining population is often older, causing shortages of suitable labour for the traditional land management practices such as heather burning. However, these changes are not always viewed negatively. Less intensive management may impact positively on biodiversity. Shiel (2002) suggested that the sale of rural land to wealthy urban migrants could have positive environmental impacts because the 'newcomers' are likely to be more sensitive to the environmental impacts of farming.

New funding and legislation is also helping to drive moorland change. The 1992 Rio Summit resulted in a UK Biodiversity Action Plan (BAP) which outlined steps to redress historic wildlife losses, and aimed to deliver and demonstrate socio-economic benefits to local people through wildlife conservation and economic incentives for wildlife-friendly farming. English Nature set targets for improvement in the ecological quality of many Sites of Special Scientific Interest (SSSI) and this has led to obligations for landowners of SSSIs (English Nature, 2003). Similar arrangements exist throughout the UK. For example in Scotland over 12% of land is designated by the devolved government conservation body as SSSI's (Scottish Natural Heritage, 2003), with SSSI protection operating through a list of Potentially Damaging Operations which require prior consultation with Scottish Natural Heritage (Reid, 1993). EU funding has also been made available for training, equal opportunities, social exclusion issues and combating unemployment (European Social Fund) and to support farmers wishing to diversify into activities such as tourism (European Agricultural Guidance and Guarantee Fund) (Arnold-Forster, 2002). This has taken place in parallel with shifts away from exclusively 'agricultural' development, towards a more holistic 'rural development' encompassing social and economic as well as agricultural needs. Changes to the farming subsidy system are currently progressing with the reform of the EU Common Agricultural Policy (CAP) system. Output-based subsidies are gradually being replaced by Single Farm payments for 'environmentally sensitive agriculture'. This rewards farmers for using more sustainable practices and promoting wildlife habitat (Lowe et al., 2002).

Furthermore, moorland managers are assessing the implications of the EU Water Framework Directive (WFD). This requires inland waters to achieve 'good ecological status' by 2015 (i.e. good chemical, morphological and biological status). Significant changes in management practices will be required to deal with non-point-source pollution (e.g. from fine organic sediment release, fertilisers and pesticides) on a catchment-wide basis. This is particularly important because there have not previously been any legal instruments requiring the control of diffuse pollutants. Landowners will, in future, be required to take action to ensure that diffuse pollution of water meets WFD standards. In UK uplands, water discoloration is a major issue since moorlands, particularly when degraded, tend to produce more discoloured water with higher concentrations of dissolved organic carbon (Driscoll et al., 2003). This is not only a WFD problem but one for raw water treatment because chlorination of highly-coloured water releases trihalomethanes, which are potentially toxic and carcinogenic (Kneale and McDonald, 1999). This may have an economic impact for water companies, and provide another driver of land management.

2.2. Atmospheric deposition

There have been significant changes to atmospheric deposition chemistry across the UK over the past 250 years. These have mainly been caused by industrial and vehicular emissions. The major atmospheric pollutants are acidifiers such as sulphur dioxide (SO_2), and nitrogen oxides (NO_x), toxic substances such as ozone, volatile organic compounds (VOCs) and heavy metals, and fertilizing substances: ammonium (NH_4) and NO_x. The deposition of these pollutants have been affected by industrial changes, agricultural productivity and the introduction of new legislation to reduce emissions (e.g. the 1999 Gothenburg Protocol which set emission ceilings for 2010 for sulphur, NO_x, VOCs and ammonia, and the Aarhus Protocol of 1998 which committed the UK to the reduction of heavy metal deposition to below 1990 levels). Historically the major acidifying pollutant has been SO_2 emitted from fossil fuel combustion. Acidic deposition is generated from the oxidation of SO_2 and NO_x to H_2SO_4 and HNO_3, respectively, and is the main source of acid rain. Acid deposition, which peaked in the early 1980s, has had a wide range of impacts upon the environment, including soil and vegetation, which have recently been comprehensively reviewed by NEGTAP (2001) and, hence, only the major points will be summarised here with relation to upland moorlands.

The impact of acidification on moorlands is highly variable, depending on the initial vegetation, soil buffering capacity and concurrent management practices. However, the dominance of peaty and base poor soils makes these moorland ecosystems particularly vulnerable to acidic deposition. Acid deposition has been linked to major changes in species composition in moorland environments (Lee et al., 1993) and widespread loss of Lichen species (NEGTAP, 2001). Where acidic deposition has been very high, such as the southern Pennines, *Sphagnum* mosses have been almost eliminated. Field experiments with *Sphagnum* have shown that it quickly succumbs to the application of SO_2 in solution (Ferguson and Lee, 1983). The loss of *Sphagnum* cover through acid pollution combined with overgrazing has been blamed for initiating erosion in a number of locations, including the southern Pennines, although broad generalisations on the influence of sulphur or nitrogen deposition on *Sphagnum* species still require further work.

Over the long-term, acidic deposition leads to a decline in soil acidity as it has the effect of increasing the leaching of base cations, such as calcium and magnesium, which leads to a decline in the proportion of cation exchange sites occupied by base cations and hence a increase in exchange sites occupied by aluminium species and H^+ ions. A decline in soil pH increases the solubility of heavy metals in the soil, such as aluminium, manganese, lead, cadmium and zinc, which can be toxic to plants (NEGTAP, 2001). This can lead to decreased plant growth or changes in plant communities. The populations of soil organisms may also change, with a shift towards more acid tolerant species. As a result, a number of soil processes can slow down. For example, the decomposition of litter becomes slower, leading to surface accumulation (Sanger et al., 1994) and slower nutrient cycling. Soil acidification gradually leads to acidification of waters draining from them (Cresser and Edwards, 1987). Acidity and high concentrations of aluminium can lead to deterioration in the ecology of streams draining moorland areas. For example, the diversity and size of invertebrates and fish populations decline (Weatherley and Ormerod, 1987; Weatherly et al., 1990).

Over the last two decades, sulphur deposition in the UK has declined by 60% (Fowler et al., 2005), making it perhaps the largest 'environmental change' across the UK moorlands in recent time. While the response of surface waters to this decline in sulphur deposition has been monitored via the Acid Water Monitoring Network (AWMN) (e.g. Davies et al., 2005), there has been no systematic monitoring of how soil biota and soil processes, such as decomposition and nutrient cycling, have responded to this large reduction in sulphur and H^+ ion inputs, and the subsequent impact on moorland vegetation and ecology.

While sulphur deposition has declined rapidly over the last 20 years, atmospheric deposition of reactive N compounds have increased and now reached levels of 40 kg N ha^{-1} $year^{-1}$ over large areas of the UK (NEGTAP, 2001). The increased N deposition onto moorlands represents a particular threat as semi-natural plants in these environments have adapted to very low levels of N inputs and occur on soils of low nitrogen supply. Hence a large amount of research has been undertaken to determine the impacts of elevated nitrogen deposition on semi-natural ecosystems (INDITE, 1994; NEGTAP, 2001). Responses range from changes in plant community composition, altered patterns of plant growth, soil acidification, changes in carbon and nutrient cycling, and deterioration in the chemical and biological status of freshwaters. A summary of the main findings are presented here.

Soil N availability is strongly associated with plant species composition in semi-natural systems (e.g. Pastor et al., 1984) and evidence from competition experiments (e.g. Aerts et al., 1990) shows that increases in N deposition over the last few decades have led to the encroachment of grass species into areas once dominated by heather (NEGTAP, 2001), although inappropriate land management practices may also play a part. Increased rates of N deposition have also been shown to lead to a decline in the cover of mosses and lichen in upland *Calluna* heaths (Carroll et al., 1999), alterations in root and shoot growth (Carroll et al., 1999), decreased species diversity (Carroll et al., 2003) and increased susceptibility to a range of environmental stresses, such as frost and drought (Caporn et al., 1994; Carroll et al., 1999).

Soil processes have also been shown to be altered in a number of ways as a result of increased N deposition, including effects on soil pH, mineralization and nitrification rates (Fog, 1988), increased carbon accumulation (Evans et al., 2006) and increased leaching of NO_3 (Chapman and Edwards, 1999; Pilkington et al., 2005b). Soils may be acidified as a result of (1) the displacement of base cations by ammonium accumulation (White and Cresser, 1998), (2) the loss of base cations from the soil through leaching with nitrate (INDITE, 1994) and (3) increased uptake and accumulation of base cation in plant biomass due to increased N inputs stimulating plant growth with consequent increased demand for nutrients and base cations (INDITE, 1994). The fertilisation effect of nitrogen deposition also increases the N concentration of plant tissue and litter (Pilkington et al., 2005a). Plant litter is the principle carbon resource reaching the soil surface and its N content and C:N ratio has been used as a guide to its quality and rate of decomposition (Heal et al., 1997). From measurements of increased N concentrations and reduced C:N ratios in tissues of mosses on peat bogs in Sweden, it has been predicted that decomposition of these materials will increase (Aerts et al., 1992). This will lead to an increase in available N for plant uptake or leaching. In contrast, others have shown that increased N deposition leads to increased recalcitrance of N-enriched litter leading to reduced long-term decomposition rates (e.g. Berg et al., 1998; Franklin et al., 2003). The impacts of direct additions of N on the decomposition of organic matter have been comprehensively reviewed by Fog (1988); both positive and negative correlations between N addition and rates of decomposition have been reported.

Sustained long-term inputs of N can lead to 'nitrogen saturation', defined as the point when N availability is greater than the combined plant and microbial demand (Aber et al., 1989) and is identified by an increase in nitrate leaching from the system. This can in turn lead to acidification and eutrophication of freshwaters. A review of input–output budgets for a number of UK upland catchments identified that moorland catchments have the ability to retain large amounts of atmospherically deposited N (Chapman and Edwards, 1999). The large organic matter pools associated with the peat and organo-mineral soils that dominate moorland areas, provide a potentially large sink for atmospheric N deposition, potentially slowing the rate of N saturation in these systems. Further research is required to determine exactly how much N these systems can retain and where it is retained in these systems. Recent work by Pilkington et al. (2005a) have shown, in an experiment designed to investigate the long-term impact of N addition to a moorland, that 90% of added N was found in the soil, particularly the organic horizon.

Recent data show that European emissions of oxidized nitrogen compounds have declined over the last decade (NEGTAP, 2001) and in the UK they have declined by

36% over the period 1986–2001 (Fowler et al., 2005). Under the Gothenburg Protocol and EU National Emission Ceiling Directive, a further reduction in emissions of oxidised nitrogen will occur by 2010, leading to a further reduction in atmospheric nitrogen inputs to terrestrial ecosystems. In the UK, terrestrial ecosystems have experienced over 50 years of atmospheric nitrogen deposition significantly above pre-industrial levels (Fowler et al., 2004), and have responded to the cumulative effects of this loading. There is very little information available to indicate the speed at which ecosystems, including moorlands, might recover as rates of nitrogen deposition begin to fall. The accumulation of nitrogen stores in litter and soil layers of recent upland heathland manipulation experiments (Pilkington et al., 2005a) suggests that, in the absence of management options targeted at removing these stores, the effects of elevated nitrogen inputs will persist for many years. Indeed, Power et al. (2006) found that *Calluna* canopy development, phenology and drought sensitivity were still affected by earlier nitrogen treatments, up to 8 years after nitrogen additions ceased at a lowland heathland and that management options, including burning, had only limited impact on the speed of recovery to pre-treatment conditions, which suggests that recovery will be a relatively slow process.

Changes in atmospheric deposition chemistry are occurring at the same time as changes in other biotic and abiotic factors, such as land management and climate. Thus, future research needs to consider the interactive effects of changes in all these environmental factors on upland moorland ecosystems. For example, a combination of field, turf and plot experiments showed that N addition to moorland vegetation only results in a transition from *Calluna* to a grass-dominated sward where the heather canopy was opened by grazing (Alonso and Hartley, 1998; Alonso et al., 2001). Effects of N are also interactive with those of temperature, but the outcome of the interaction is difficult to predict since it varies between species and environments (e.g. Jonasson et al., 1999) and thus requires further research. Finally, prolonged deposition to low nutrient ecosystems can result in phosphorus (P) limitation, such that further response to N is limited by P availability (e.g. Aerts et al., 1992).

2.3. Climate change

Climate change has always played an in important role in moorland ecosystems. Many species tend to have altitudinal limits related to temperature, or sunlight exposure and these are likely to change as annual means and seasonality of both temperature and precipitation in the uplands are altered over the next century (Holden and Adamson, 2002; Burt and Horton, 2003). The current rate of anthropogenic climate forcing is a particular cause for concern in moorland environments. Temperatures across the UK moorlands are likely to increase by 0.8 to 2 °C by 2050 (Tallis et al., 1997) although most of the recent upland warming has been confined to the winter months with associated decreases in lapse rates (rate at which temperature declines with altitude) (Pepin, 1995; Holden and Adamson, 2002). Temperature increases are not the only predicted change due to climate change. Precipitation totals are predicted to increase in the north and west, with a stronger winter–summer contrast (Burt et al., 1998), but to decrease in the south and east, enhancing existing environmental gradients and placing the most southerly and easterly moorlands under more extreme pressure and potentially increasing runoff in many upland areas (Werrity, 2002). Furthermore, changes in seasonality of rainfall and temperature will lead to increased frequency and/or severity of summer drought in northern Britain (Worrall et al., 2006).

With climate change species could be expected to migrate. This migration is amenable to monitoring and can be modelled to provide predictions (Huntley and Baxter, 2002). However, where moorland environments are fragmented (e.g. by afforestation) or destroyed, the capacity for migration is severely reduced and extinctions might be expected. This effect is greatest when the remaining habitat is in discrete but more isolated patches, rather than scattered through a fine-grained landscape (Huntley and Baxter, 2002). These results, therefore, have implications for conservation planning. Conservation bodies have previously focussed attention on small reserves, often responding to *ad hoc* decisions or opportunity (Gaston et al., 2006) and not the wider landscape in a holistic approach. However given the expected changes to anthropogenic forcing, a more strategic approach may need to be adopted to allow for change and migration.

Peatlands represent the most significant terrestrial carbon pool in the UK (Cannell et al., 1993; Milne and Brown, 1997). Though they form a significant reserve, they can be both sinks and sources of carbon (Shurpali et al., 1993). The evidence to date from UK peatlands suggests that they are probably a slight net carbon sink (Worrall et al., 2003b), but this could change to a net source under future climate change (Worrall and Burt, in press). The dominant controls on the peatland carbon cycle are often stated as plant community, temperature, water table position, and the chemistry of the peat. Empirical relationships have been developed to examine the release of CH_4 and CO_2 from the peat surface (e.g.

Lloyd and Taylor, 1994), and the decomposition of peat into dissolved organic carbon which is then released in runoff (Worrall et al., 2005). From these relationships global estimates of current carbon emission from peatlands are produced and predictions made (Davidson and Janssens, 2006). However, none of these approaches combine all carbon uptake and release pathways in order to obtain a complete carbon budget for the particular ecosystem. However, given that air and soil temperatures will increase in the future it can be expected that all reaction rates within the peat will increase. Of particular concern would be the balance between the rate of uptake processes and the rate of decomposition processes. Under normal peat temperature ranges, CO_2 production increases by threefold for every 10 °C increase, but this varies with depth and it is not clear what controls the temperature dependency of carbon mineralization rates (Blodau, 2002). Equally, there is evidence that losses of dissolved organic carbon will increase with increasing temperature (Freeman et al., 2001).

In contrast, many moorland soils such as peats isolate carbon from atmospheric CO_2 through plant photosynthesis and this reaction is controlled not by temperature but by the amount of incident radiation (e.g. Bubier et al., 1998). However, climate change is not solely about increased temperature, and the underlying cause of increased air temperatures is enhanced atmospheric CO_2. Elevated CO_2 has been shown to increase primary production (e.g. Gill et al., 2002) and greater carbon input is expected to increase carbon sequestration in soil by this mechanism. Other evidence suggests that the response of plants to elevated CO_2 in peatlands is very mixed, with some studies observing no increase in biomass growth under elevated CO_2 (e.g. Berendse et al., 2001) and others a change in plant species composition (e.g. Freeman et al., 2004) with the abundance of vascular plants increasing relative to mosses. For other carbon pathways increases in atmospheric CO_2 could lead to increased losses of carbon rather than increased storage. Freeman et al. (2004) observed an increase in dissolved organic carbon from peat soils under elevated CO_2 which they attributed to elevated net primary productivity, and increased root exudation of dissolved organic carbon. They suggest that the labile carbon released by roots stimulate microbial activity, leading to enhanced degradation of soil organic matter; this process is known as the ‘priming mechanism’. Hence Freeman et al. (2004) suggest that the increase in CO_2 is responsible for the increase in concentration of dissolved organic carbon observed in freshwaters across large areas of Europe and North America (e.g. Driscoll et al., 2003; Worrall et al., 2004; Evans et al., 2005; Skjelkvåle et al., 2006). The potential environmental implications of the increase in dissolved organic carbon are wide ranging, from local effects on water transparency, acidity and metal toxicity through to effects on drinking water quality, and possible destabilisation of terrestrial carbon stores, increasing fluxes into more reactive (riverine, marine and ultimately atmospheric) pools. Several other possibilities have been proposed: increasing air temperature (Freeman et al., 2001); changes in land management (Worrall et al., 2003a); change in the amount and nature of flow (Tranvik and Jansson, 2002); eutrophication (Harriman et al., 1998); recovery from acidification (Evans et al., 2005), and the action of severe summer drought (Worrall and Burt, 2004). At present the exact processes causing the increase in dissolved organic carbon export from peatlands are unclear.

The change in air temperature and precipitation regimes with climate change will mean increased depths to water table in organic moorland soils. Many organic soils, such as peats, convert some sequestered carbon anaerobically into CH_4 which is much more potent as a greenhouse gas than CO_2. If the water table is lowered, the carbon sink–source relationship is likely to be disturbed because a greater percentage of the peat is available for oxidation in biochemical reactions. In addition, the rate of peat decomposition will increase with lowered water tables, and effectively more CO_2 and dissolved organic carbon will be available for release. However, as a potential counter-balance, reduced water tables would result in a reduction in the concentration of CH_4 released because the increase in aerobic conditions will suppress the activity of the anaerobic methanogenic bacteria and increase the volume of peat in which CH_4 oxidation may occur. However, Hughes et al. (1999) artificially drained a moorland soil to simulate climatically driven changes in water table and found that CH_4 emissions demonstrated a 3 year cycle (large increased emissions, then very low emissions and then recovery back to pre-drainage state).

Change in precipitation and temperature regime leading to increased drought frequency and or severity can have further additional effects upon carbon storage in peats. Freeman et al. (2001) have shown that hydrolase enzymes in peat bogs are inhibited by the presence of phenolic compounds, which can build up in peat because the activity of phenol oxidase is severely restricted in the absence of oxygen. Therefore, if the water table in peat bogs falls, the phenyl oxidase activity increases and oxygen ingress increases, destroying the phenolic compounds that repress the hydrolase activity. A loss of phenolic compounds means that decomposition can continue even after the water table has risen again. Droughts could augment the dissolved organic carbon production by causing a drop in the water table below the long-term average position (the acrotelm–catotelm boundary, Holden and Burt, 2003b) and triggering an additional anaerobic production. There are several lines of evidence to support this mechanism (e.g. Worrall and Burt, 2004). However, Worrall et al. (2005) found that changes in dissolved organic carbon production after a severe drought had no effect on the carbon release pathways. An alternative drought mechanism was proposed by Clark et al. (2005). The catastrophic lowering of the water table in peat during droughts leads to the oxidation of sulphide minerals to sulphate. The increase in sulphate concentration suppresses the mobility of dissolved organic carbon, as the drought ends this suppression is released and the dissolved organic carbon concentrations rise. Worrall and Burt (in press) have examined national records of dissolved organic carbon and found that drought effects could only explain 7% of the observed variation in data, while temperature increase could explain 17%.

Climate change in moorlands will also alter the water balance and hence change amount and/or the nature of runoff. Tranvik and Jansson (2002) and Worrall and Burt (in press) have both ascribed much of the observed variation in dissolved organic carbon to changes in the amount of runoff from peat-covered catchments. However, for other types of carbon release runoff changes may alter the release rates and pathways (e.g. Sanger et al., 1994; Haines-Young et al., 2000).

2.4. Stakeholder priorities

A series of discussions were held between the authors of this paper and moorland stakeholders in 2005 and 2006 as described in Dougill et al. (2006). These revolved around consideration of the various drivers of change and attempted to ascertain the most important factors that stakeholders felt were likely to change in the uplands in response to these drivers. The most important factors that consistently emerged from stakeholder meetings were:

1. There is likely to be continued 'restoration practice' such as blocking of moorland drains and moorland gullies;
2. Moorland burning practice is likely to change;
3. There is likely to be further reductions in the number of sheep;
4. There is likely to be major changes in afforestation practice;
5. Atmospheric deposition changes and climate change will continue to impact moorland processes (e.g. increased risk of wildfire).

All of these factors were considered to be major causes of anxiety and concern for stakeholders and were a significant cause of tension. The first four are responses to drivers of change. Many stakeholders pointed out that more scientific evidence is needed to determine how moorland systems respond and can be best managed under certain scenarios of change. One stakeholder even noted during a public meeting that historically, when a policy or social driver leads to a different form of moorland management, scientists tend to come along a few years later to try to examine the impacts of the new management. This was, in his view, a mistake, and the stakeholder suggested that research that could drive policy forward should be funded first, rather than allowing policy to shape subsequent research. Another stakeholder took this point further, calling for policy-makers to consider local knowledge alongside scientific evidence to develop land management policies that can stand the test of time:

"No one on the conservation side has explained to me yet why their view of the world will be anymore correct (whatever correct is) than the Forest Commission's was in 1976 when we were all taught to…plough up heather moorland, and yet everybody now assumes that they're right… I've spent thirty years managing land and I've seen all these things come and go. So when you tell me as a very sincere young man with a great deal of credentials, that your prescription is right, you just listen to me: the guy who gave me 100% grant aid…to plough heather moorland also believed he was right because heather moorland was "waste". "Why keep heather moorland? Why not grow Sitka Spruce on it?" They weren't all liars and cheats and thieves and incompetents. That was not the case. And they all look at you in absolute amazement."

Anonymous grouse moor agent

The remainder of this paper will, therefore, examine each of the above key responses to drivers of change and review the scientific evidence for likely impacts on moorland processes.

3. Responses to drivers: changes in land management

3.1. Gully and ditch blocking

The UK is one of the most extensively drained lands in Europe (Baldock et al., 1984) and drainage has played a fundamental role in the history of British farming (Avery et al., 1995), with important environmental consequences. Until the 20th century most land drainage was focussed on ‘improving’ lowlands for agriculture by lowering the water table. It was in the 1960s and 1970s that most of the moorland drainage took place, particularly in the English Pennines. Partly this drainage was to improve the quality of grazing and partly to remove the hazard to stock (Ratcliffe and Oswald, 1988). However, Stewart and Lance (1983) demonstrated that there was no evidence that peatland draining fulfilled the claims made for it. Grouse populations did not seem to increase because of drainage and while drains were the cue for increases in stocking density there was little evidence that the moors could sustain large increases.

Moorland drainage has been associated with environmental degradation. The drainage resulted in changes in water flow paths through and over moorland soils (Holden et al., 2006a). Both increases and decreases in flood peaks have been observed (Holden et al., 2004). This is because while water table lowering buffers (slightly) the impacts of a rainfall event by providing extra soil storage capacity for rainwater and reducing saturation-excess overland flow in the early stages of a storm, higher flow velocities in the ditches themselves speed up the delivery of water from the land into streams, resulting in a complex hydrological response. Many drained peatland catchments exhibit increases in low flows (Baden and Eggelsmann, 1970; Robinson, 1985). This has often been attributed to catchment ‘dewatering’ following drainage (Burke, 1975) and changes to soil structure (Holden, 2005a). Moorlands are typically underlain by highly organic soils that shrink and crack (Holden and Burt, 2002a,b) and decompose when dried. This change in soil structure is important for hydrology, water quality and ecology in moorlands.

Many moorland catchments have a high proportion of soil pipes running below the surface (Holden and Burt, 2002c). Soil pipes are natural subsurface channels that transport water, sediment and solutes through moorland hillslopes. Jones and Crane (1984) found that around 50% of streamflow moved through soil pipes in a shallow peaty podzolic moor on Plynlimon, mid-Wales, while Holden and Burt (2002c) measured around 10% of streamflow moving through blanket peat soil pipes. Pipes can often produce greater amounts of sediment than the hillslope surfaces (Jones, 2004) and are therefore important for river water quality and carbon release. Holden (2005a) found that moorlands that had been drained had significantly higher amounts of soil piping than other moorlands. In blanket peats Holden (2006) has shown that, as the drain networks get older, the density of piping increases and the pipes enlarge. This response to drainage (which can continue even 80 years after drainage) results in an exponential increase in sediment (or particulate carbon) release from the soil and long-term change in river flow (Holden et al., 2006a). Drain blocking should prioritise the older drains, if sediment and carbon release is considered to be a significant problem. In addition to sediment release from soil piping, drainage ditches themselves can be subject to severe scouring, widening and deepening, often by several metres (Mayfield and Pearson, 1972). Site characteristics (e.g. steep slopes) often mean that even recently drained catchments may be significant sources of sediment and carbon. The deepened gullies can become a hazard for stock and humans and the eroded organic sediment can cover gravel bed spawning grounds downstream and infill reservoirs. Although little is known about impacts of this sediment on stream ecosystems, an unpublished sediment survey for three moorland basins in the North Pennines showed that drains generated over 60% of the organic sediment in the stream system from only 8% of the area.

One of the hydrological effects of moorland drainage is to lower the water table, thereby increasing the air-filled porosity of the peat. This affects microbial processes and increases decomposition rates. Access to oxygen from the air allows aerobic decomposition to take place, which occurs at a rate about fifty times faster than anoxic decomposition (Clymo, 1983), and enhances the mineralization of nutrients, particularly carbon-bound nitrogen and sulphur and organically bound phosphorus. Even a small increase in the mineralization of just one per cent per year has the potential to generate significant additional losses of carbon, phosphorus, nitrogen and sulphur and may in turn affect the fertility of peat. Many studies have observed that excavation of drainage ditches usually increases the leaching of nutrients. For example, large increases in ammonium (NH_4) concentrations within the streamwater have been observed in the decade following drainage (Lundin, 1991; Sallantaus, 1995; Miller et al., 1996) and during droughts which cause water table lowering (Ross et al., 2000), but only small changes in nitrate (NO_3) concentrations. This suggests that while the organisms for ammonification benefited from drainage, those responsible for nitrification did not do so to the same extent. However, increased NO_3 and base cation losses have been reported from less acidic peats (Burt et al., 1990; Lundin, 1991; Freeman et al., 1993). Sallantaus (1995) observed a net loss of calcium, magnesium and potassium from drained catchments compared to undrained catchments, where inputs and outputs of these nutrients were more or less balanced. Drained peat soils have been found to contain more humus compounds and substances which are readily hydrolysed and hence Mitchell and McDonald (1995) and Clausen (1980) found that drained catchments produced much more discoloured water than undrained catchments. While this presents a general picture, the effects can depend on the catchment characteristics so that in some locations only minor increases or even significantly lower concentrations of dissolved organic carbon (associated with water discolouration) have been observed in streams flowing from drained catchments compared to nearby intact moorlands (Moore, 1987; Chapman et al., 1999; Driscoll et al., 2003). The temporal pattern is also important. Hughes et al. (1997), for example, artificially lowered water tables in a small wetland in Wales and found that this decreased summer dissolved organic carbon and acidity peaks while increasing autumn–winter peaks. It should be noted that the majority of studies of the impact of artificial ditching on water chemistry have observed changes for less than five years, with a dearth of longer term studies.

In some UK moorlands gully erosion is a major problem. The problems associated with gullying are the same as those described above for drainage, but they tend to be more severe. Gullies tend to provide a deeper, more wandering and branching network of channels (see Fig. 2). Gully erosion tends to be most severe on very low gradient interfluves (Bower, 1960) and the intricate channel network draining hummock and pool topography can greatly increase storm discharge (Conway and Millar, 1960; Burt and Gardiner, 1984). In a drive to reverse the degradation caused by moorland drainage and gully development, many moorland areas now adopt a policy of blocking ditches and gullies. However, it may not be a simple task to reverse these disturbances because changes to soil pH, nutrient status and soil structure (e.g. soil pipe development) as a result of gully incision can make ecological restoration difficult.

Moorland restoration often involves raising the water table. If natural revegetation does not then ensue, or if it is deemed unlikely in the first instance, active measures are pursued, often through reseeding and application of a mulch (Campeau and Rochefort, 1996; Price, 1997) or heather brash (Evans et al., 2005) to stabilise the surface and retain surface moisture. Drain and gully blocking may not be sufficient to restore the water table to its original mean height or its natural range of fluctuation (both of which are important for sensitive moorland plants) and management practices may then instead be aimed at reducing the rate of degradation and sediment loss, rather than total restoration of the moorland ecosystems. Nevertheless, the goal of most blocking activity is to ‘restore’ the water table and hydrological regime to a former, more ‘pristine’ state.

Gully and drain blocking may be a daunting prospect for moorland managers. This is because there are hundreds of thousands of kilometres of land drains and gullies that could be blocked across the UK alone, and much more across Europe. Over the past few years most blocking has been done on an *ad hoc* and piecemeal basis and many blocking styles that have been trialled (e.g. using plastic piling, heather bails, wooden dams) have proved enormously expensive. Evans et al. (2005), for example, suggested that a plastic dam in a small gully might cost up to £50 plus labour costs, however, there are also cheaper alternatives using such as using wood or heather bales, for which material costs are much reduced. These dams are placed every few metres along gully channels. We calculate that to block small ditches in only a 13 km^2 area of Upper Wharfedale, northern England would cost around £100k if plastic dams or heather bales were used. An important cost factor is the helicopter time to carry material to remote places. Given that such management activity will continue, it is necessary to determine ways of more cost-effectively implementing the blocking solutions (e.g. where and how to block), and to understand what the impacts of blocking might be. Furthermore, it is important to set the objectives, of whether to reduce erosion or raise the water table and this determines the choice of material (e.g. choosing permeable or non-permeable blocking material, such as heather bales or plastic).

There are some recent advances in understanding potential impacts of blocking and in prioritising methods and locations. Wallage et al. (2006), for example, have shown that dissolved organic carbon and water colour production from a site blocked three years prior to measurement was significantly lower than the adjacent drained site, but also significantly lower than that from undrained moorland. This suggests that drain blocking is an effective treatment for water colour problems. However, Wallage et al. (2006) also found that the constituents of the dissolved organic carbon produced at the blocked site were different to that from intact sites due to changes in biogeochemical production processes. The concern is that, while the colour and dissolved organic carbon production were reduced, the type of organic matter released was more difficult for water companies to treat. This example demonstrates how moorland systems do not necessarily respond in a linear and reversible way to management drivers.

Recently, it has been realised that simple topographic models can be used to determine which land drains or gullies are more important in reducing the saturation downslope in moorlands. It has been demonstrated that slope is an important factor in determining drain erosion and therefore, to be effective, drain blocking should be concentrated on those on slopes over 4° (Holden et al., 2006b). Furthermore, if a drain runs across a slope, it will intercept flow from upslope and prevent it flowing downslope, so that reduction in water table is greatest downslope of the drain. By mapping the drainage area and slope characteristics it is therefore possible to determine which individual land drains or gullies should be targeted for blocking (Clark et al., 2005; Evans et al., 2005; Holden et al., 2006b). During our own work no significant difference has been observed between the water table recovery or vegetation change in drains blocked by expensive methods and by drains blocked using simple peat dams. The caveat to this, however, is that peat blocks cannot be used on steep slopes (and usually not in gullies) and have to be carefully designed. They should be firmly squeezed into the ditch floor and packed, and there should be an escape route for water away from the ditch channel (Fig. 3) so that the hillslope surface re-wets again. The additional caveat is that grazing and burning should also be managed appropriately, if investment in blocking is going to be effective (see below).

Gully and ditch blocking are likely to continue in UK moorlands over the next decade. This will be involve a large financial investment but should reduce sediment loads and decrease water discolouration. It is possible to use the existing data to predict the magnitude of change in sediment and water colour production that will be brought about by block management for the given catchments (Worrall et al., 2005). However, while vegetation recovery can be rapid in the immediate vicinity of a blocked drain or gully, it may take several decades for the impacts to be measurable at larger distances from the blocked channels. It is also not known how blocking will impact on other water quality or soil processes. Given that many drained peats have much larger densities of soil pipes, it is important that these pipes are considered in management solutions. It may be that damming drains simply allows more water to enter pipe networks that have openings on ditch floors and sides. More use should be made of geomorphological evidence to prioritise blocking. Spatial and topographical analyses should always be performed to make sure that any investment on moorland drain or gully blocking is cost-effective and will provide the maximum area with a potential for 'restoration'. At the moment, most drain blocking is occurring in the same vein as drain creation did in the 20th Century; without consideration of natural processes and of the importance of understanding the role of each site in terms of its local setting and within the catchment as a whole.

Fig. 3. Ditch blocking using a carefully designed peat dam. *Sphagnum* carpets often develop where water is allowed to escape back onto the hillside.

3.2. Moorland burning

Moorlands have traditionally been burned to manage heather (and sometimes grass) for sheep and deer (in large expanses to favour young heather shoots for winter fodder, and palatable sedges and grasses) and red grouse (in small rotationally burned patches). While a mosaic of woodland, scrub and dwarf-shrub heaths replaced much of the native woodland cleared by humans in the mid to late Holocene, the advent of treeless, rotationally burned grouse moor was relatively recent. Since the early 1800s in England and about the 1840s in Scotland, heather has been burned in rotation to produce very high densities of red grouse. The practice of moorland patch burning seems to have originated following a report by Lovat (1911) who recommended that the area to be burned should depend on the time taken for the heather to recover and grow back to its desired height on typically 8 to 25 year rotation. Lovat advocated narrow strips but stressed that the desire to burn small patches should not take precedence over burning sufficient areas to maintain the required burning rotation. In the 1960s and 1970s a series of other papers were produced examining optimum sizes and rotation periods for burn patches (e.g. Gimingham, 1971, 1972). It was shown that on heather moorland, if the heather is left for too long, then other dominant species such as *Molinia* start to take over, the heather may burn too fiercely when not managed properly, and there is an increased risk from natural or accidental wildfire (Kenworthy, 1963). If burned too frequently, then the heather can be lost (Grant, 1968). Local factors were found to be important (Legg et al., 1992) and the length of time between burns was recommended to be longer on blanket bog (Miller et al., 1984). Sometimes burning does not encourage enhanced heather growth. For example, Ross et al. (2003) have shown that for moors where there is a combined heather and *Molinia* dominance, burning of the moorland increased *Molinia* cover at the expense of the heather. Therefore, burning *Molinia*-dominated heaths is not recommended as a management practice (Scottish Natural Heritage, 1993; MAFF, 1994). Since the 1970s there has been a gradual decline in grouse populations despite continued burning and despite the feeding of medicated grit to grouse (Simmons, 2002). This illustrates how burning is only one of the many factors affecting wildlife populations in the moorlands. The lack of profitability of some grouse moors, lack of trained rural population to carry out the right type of burns, the environmental concerns around burning and the policy shifts towards more holistic, multi-purpose management goals suggest that perhaps it is an appropriate point in time to re-evaluate these management practices.

Burning in England and Wales is currently regulated by the Heather and Grass Burning Code (MAFF, 1994) and in Scotland by the Muirburn Code (Scottish Natural Heritage, 1993). These codes provide legal dates between which burning is allowed and also guidelines on burning practice. The Heather and Grass Burning Code allows burning between 1st November and 31st March in lowland areas and between 1st October to 15th April in the uplands. The autumn limit is to prevent wildfire risk, which is greater in the dry, summer months, while the spring limit aims to protect breeding birds. Both codes provide a list of recommendations rather than instructions and there is actually no legal imperative to comply (except to dates of burning). The codes state that certain areas should not be burned including blanket bog, steep slopes where there is a risk of erosion, sites above the 600 m natural limit of forest and large areas of old rank heather. However, as an unintended response to ESA payments there has, in fact, been an increase in the area of land being burned, at least in some parts of northern England, since 1995 (Yallop et al., 2006). Air photo evidence suggests that burning has significantly encroached on to blanket bog as the economic incentive has grown for industrial scale grouse farming. At the same time, there is serious concern about the environmental implications of burning (on water quality, carbon release, protection of blanket bogs etc). These concerns mean that there are likely to be some changes in policy and legal enforcement in forthcoming years (notwithstanding the requirements of the WFD).

There have been a number of recent and comprehensive reviews on the impacts of moor burning on environmental processes (Hobbs and Gimingham, 1987; Mowforth and Sydes, 1989; Shaw et al., 1996; Tucker, 2003; Glaves and Haycock, 2005). The Glaves and Haycock (2005) review of the Heather and Grass Burning code for the UK government agency (DEFRA) noted that, because of the lack of scientific data it was difficult to provide evidence to support any major changes in the code. Virtually nothing is known about whether burning influences the moorland hydrology, sediment release and water quality. Although a substantial amount of research has been published on the effect of burning on blanket bog and dwarf-shrub health communities, there is insufficient evidence to determine its effect on floristic diversity (Stewart et al., 2004a,b). These are all experimentally determinate factors that require scientific funding and considerable detailed research.

Of the little research that has been done, the work by Holden (2005c) is notable because it has shown that heather is associated with more soil piping in moorlands and increased soil piping leads to changes in hydrological flowpaths and changes in water quality and carbon fluxes. In terms of more direct influences of burning on moorland soils, wildfire (which often burns for longer and to hotter temperatures than managed burns) has been shown to result in the development of water-repellent compounds (Clymo, 1983). The removal of vegetation can make the soil surface susceptible to wind and fluvial erosion as well as to increased freeze–thaw action. In many moorland fire erosion studies there has been a lack of careful experimental design so that results cannot be interpreted more widely. Some authors have attributed the onset of major erosion episodes to historic wildfire (Mackay and Tallis, 1996) or historic human-induced fire (Tallis, 1987). There is a dearth of data on infiltration following moorland fire but the increases (Kinako, 1975) and decreases (Mallik et al., 1984) provided some evidence. Increases in pH (Allen, 1964; Stevenson et al., 1996) are also likely, and differences in pH have been noted between different burning regimes on blanket bog (Worrall et al., in press) as have some influences of ash on microbial populations (MacDonald, 2000).

With increasing concern over carbon sinks and sources, the impact of moorland burning has come under increased scrutiny. Garnett et al. (2000) examined long-term experimental plots at Moor House, North Pennines, and found that burning reduces peat accumulation in comparison to no burning. It was suggested, in line with studies in Canada (Kuhry, 1994) and Finland (Pitkänen et al., 1999), that the cessation of moorland burning would be one mechanism for reducing carbon emissions and increasing carbon sequestration.

Other nutrients are lost in the smoke during burning, as particulate matter, and through volatilisation (over 50% of carbon, nitrogen and sulphur are lost from heather for example; Allen, 1964). The concentrations of soil nutrients tend to be high for the first two years after a burn, benefiting regeneration (Hansen, 1969). However, leaching may be significant particularly after autumn burns. Losses of phosphorus and nitrogen may impact future vegetation growth and Kinako and Gimingham (1980) suggested that it may take 75 years for the phosphorus losses from one burn to be replaced. There are therefore questions about whether burning is leading to a long-term decline in moorland productivity (Coulson et al., 1992) or whether it is actually preventing an accumulation that would otherwise lead to a change in vegetation community towards trees, grasses and bracken (Gimingham, 1995). Recent increases in atmospheric nitrogen deposition onto moorlands, that increase nutrient status have been implicated as one reason for the widespread shift from heather to grasses or sedges in parts of the UK (Ross et al., 2003). There may be a role for burning in reducing N accumulation: management burns have been found to remove a significant amount of nitrogen, however, subsequent regrowth of heather was inhibited in high nitrogen treated plots (Ashmore et al., 2003). The influence of burning on the long-term cycling of N in moorland soils is at present unknown.

It is generally accepted that some degree of management is usually required for the perpetuation of the status quo on fairly dry heather moorlands, but appropriate measures must be carefully considered, as the balance between the main plant species is extremely sensitive, and much depends on the climate, soil type and drainage (Shaw et al., 1996). It is clear that for many moorlands burning is highly effective in keeping heather at a productive stage, where the seed bank and vegetative regeneration potential is at the highest. However, it should be noted that we do not sufficiently understand the role of local burning histories in the production of different moorland communities. This makes it difficult to establish modern ‘good management practice’. The prediction of how vegetation communities would develop in the absence of regular burning is also difficult because the pattern, speed and outcome of secondary succession processes varies greatly from place to place (Tucker, 2003).

For blanket bogs there is a mounting literature calling for a ban on burning (Coulson et al., 1992; Usher and Thompson, 1993; Shaw et al., 1996; Tucker, 2003), particularly in catchments that are used for potable drinking water supply (although there is still a lack of evidence about the impacts of moorland burning on water quality). However, the exact definition of blanket bog is not clearly established as it is simply one end of the moorland continuum. Certainly a farmer’s definition of a blanket bog (and hence whether that farmer decides to burn it) often differs from that provided by conservation agencies. The ban on burning of blanket bog has been argued for a number of reasons, including the importance of blanket bogs to the global carbon store. There is also evidence to suggest that heather on blanket bog does not exhibit the cycle that has been identified on dry heather moorland because stems are buried by layers of *Sphagnum* as it builds up over time and the heather plants push out new shoots. This burial will only occur, however, in active peat bogs: (i.e. undrained bogs) with high water tables that prevent decomposition of the peat. Therefore, burning may not be required to provide rejuvenation of the bushes (Mowforth and Sydes, 1989). If blanket bogs are burned, often *Eriophorum vaginatum* dominates temporarily after fire and can assume permanent dominance if the community is burnt frequently (Rawes and Hobbs, 1979; Hobbs, 1984). DEFRA’s ongoing review of the Heather and Grass Burning Code in England and Wales reflects calls from conservationists for a ban on managed burning of blanket bogs. An outright ban on burning blanket bog is one option that is being considered as part of this review. Tighter controls (falling short of a ban) are being considered on other moorland types (e.g. ensuring smaller and cooler burns and a reduction in total burning). However, moves to tighten regulations are opposed by many owners and managers of heather moorland who wish to retain as much flexibility as possible (Reed et al., 2005).

There are a number of alternatives to burning if the maintenance of the heather cycle is required. Heather cutting has been trialled in some locations and would certainly have less immediate impact than burning. However, on Dartmoor in southwest England, regrowth rates of heather were slower after cutting than after burning, although in other locations there has been little observed difference (Brown, 1990). The additional benefits of cutting are that it can be done at any time of year, without impacting soil microbial processes very greatly and the cut material itself can be used to regenerate heather (or infill ditches and gullies) elsewhere. Milligan et al. (2004) found that repeated cutting (as opposed to burning) reduced *Molinia* cover and that was seen to be beneficial because *Molinia* is perceived to be a threat to heather moorland. Cutting may, however, be restricted on stony, very damp, or steep and remote terrain and is considered by many land managers to be uneconomical compared to burning (Reed et al., 2005).

Grazing could also act as a control on a moorland landscape preventing scrub development. Gimingham (1995) suggested that management of heather moorland is essential for its maintenance because heather would otherwise be replaced by other dominants. However, Gimingham also noted that nature conservation may be better served by a mosaic of stands of different age, structure and composition. One possibility is that certain areas of the landscape could be taken out of burning to allow natural succession. This would increase scrub and woodland cover and habitat diversity. However, it is clear that there is a need to manage moorland burning from a spatial perspective and so it would be necessary to determine optimum locations for such low intensity management so that maximum biodiversity, hydrological, carbon and water quality benefits might be delivered without unnecessarily increasing fire risk.

3.3. Moorland grazing

Moorlands cannot normally sustain large densities of sheep, cattle, or deer without being severely degraded. Heather tends to be unpalatable in the summer, but is of high value as a winter feed when hill grasses have died back and forms the basis of the diet for many winter hill sheep. Most heather grows only when grazing is below 2 sheep ha^{-1}. CAP subsidies in the 1970s and 1980s resulted in increased stocking so that 29% of moors were stocked above this level in 1977. By 1987 this had increased to 71%. Rawes and Hobbs (1979) found that for north Pennine blanket bogs grazing densities over 0.55 sheep ha^{-1} instigated erosion. Evans (2005) has shown that in some moorland catchments (as in many parts of the world; Evans, 1998) grazing-induced erosion is the main cause of soil degradation.

The impact of sheep grazing on erosion is most directly expressed through the formation of scars in the surface cover, where sheep shelter and rub themselves. Isolated scars lead to local movement of soil material, although most of the material is re-deposited immediately downslope. However scars are concentrated along sheep tracks and at local convexities, most significantly at the margins of old or fresh gullies, and much of the sediment from erosion of these scars is delivered directly to channels (Evans et al., 2005; Evans, 1998).

Reductions in grazing pressure can sometimes result in rapid recolonisation of eroded scars, except in many peat catchments where erosion can often continue if unchecked by human intervention. The area of moorland affected by grazing-induced erosion has increased rapidly over the past three decades (Harrod et al., 2000). In Scotland deer can cause major soil erosion but only recently has the Deer Commission for Scotland stated that it wishes to identify areas where damage by deer is occurring, and develop effective management techniques to deal with this (Price and Thomson, 2004). However, the management of Scottish deer is a somewhat different process to that of sheep. Deer are considered to be 'wild animals' and while individual landowners enjoy the rights to stalk and cull such animals, they only become their property after being killed.

In the mid 1990s, as today, the main calls for changes to moorland management were for a large reduction in sheep numbers (Marrs and Welch, 1991; Thompson et al., 1995). This reduction does appear to be occurring, mainly because of the changes to agricultural policy such as the introduction of Environmentally Sensitive Area schemes in England and Wales. It is likely that we will continue to see reductions in sheep numbers across the UK. However, deer graze over large estates in Scotland and it is unlikely that we will see a large decrease in their population. Periodically the deer population is culled but never to very low levels. The range of each individual species of deer seems to be increasing due to increased woodland habitat and milder winters.

There have been a number of studies that have examined the impact of grazing on vegetation dynamics. In terms of seed production and dispersal, grazing can have both positive and negative impacts; heavy browsing can keep saplings under severe check and prevent seed production, but consumption of seeds and dispersal in dung can be an important mechanism of spread for some species such as rowan *Sorbus* spp. and juniper *Juniperus* spp. (Thompson et al., 1995). Depending upon its intensity, grazing can reduce competitive vigour, or even kill plants through defoliation and direct damage. Overgrazing is thought to be a major cause of loss of heather moorland (Shaw et al., 1996). Grazing can have a profound effect on species composition. For example, tussocks of species more tolerant of grazing, such as *F. ovina* with *Agrostis* spp., *N. stricta*, *Molinia caerulea*, *Juncus squarrosus* or *E. vaginatum* may result according to soil type and drainage (Gimingham, 1995).

The impacts of grazing on vegetation vary with species. Sheep bite and shear vegetation to produce an even sward. Cattle, however, wrap their tongues around the vegetation and pull producing uneven vegetation. Cattle are also less selective, eat a larger proportion of *Eriophorum* spp. and will eat more *N. stricta* than sheep or deer. Both sheep and cattle tend to avoid heather, *Erica* spp. and *Empetrum nigrum*, all of which tend to be eaten mostly outside the main growing season when preferred vegetation is not available (Shaw et al., 1996). Different breeds of sheep also have different nutritional requirements. Black faced sheep can survive on mostly heather whereas some other breeds need more grasses. Although once abundant in the moorlands, there are now very few goats and they are less selective than other large herbivores. Red deer prefer a similar habitat to sheep, preferentially selecting grass sward, although they tend to eat proportionately less grass. Deer grazing can have both a detrimental impact on vertebrate and invertebrate fauna (e.g. in forests) and a beneficial impact (many rare carrion feeders and dung beetles benefit from higher deer populations). Some alpine species such as *Gentiana nivalis* require trampling and grazing to create regeneration niches, yet deer can also damage commercial forestry by stripping bark and fraying saplings. At the same time, fencing to exclude deer from parts of forests can concentrate grazing outside the exclosure, creating an excessively uniform canopy unsuitable for many woodland species.

A number of models have been developed to determine sustainable levels of grazing on moorlands. They tend to calculate the seasonal and long-term food resource available to herbivores from moorland vegetation (Sibbald et al., 1987; Grant and Armstrong, 1993; Armstrong et al., 1997). Simpson et al. (1998), for example, found that their model predicted that 15% of Orkney's heather moorland was being overgrazed and 47% for Shetland. Across the Northern Isles of Scotland a rate between 0.48 and 0.98 ewes ha^{-1} was recommended. Read et al. (2002) found that levels of sustainable grazing predicted by HeathMod were lower than previous estimates, because it was possible to provide long-term prediction of grazing impacts. Research is now beginning to indicate the need for spatially distributed models of grazing. Key vegetation attracts sheep and deer. Any neighbouring vegetation receives a higher impact than if it is associated with patches of less preferred vegetation (Palmer et al., 2004). A simple example of this is that heather near to areas of grass will tend to be grazed more heavily than more distant heather. This means that heather management decisions based on stocking density alone are insufficient because local differences in the availability of preferred vegetation so strongly influence the locations and patterns of critical impact upon heather. Modelling work has also shown that spatial interactions between herbivores and their forage drive moorland vegetation dynamics, leading to changes in community structure and composition. The comparison of spatial and non-spatial models showed that non-spatial models performed to a much lower quality and led to inaccurate predictions of heather utilisation (Palmer et al., 2004). These complex processes are one reason why it is not possible to apply a universal stocking density for the UK moorlands. Furthermore regulations and predictions regarding stocking levels have not yet taken account of the increasing trend towards the use of large sheep breeds and tend to be based solely on sheep numbers. Additionally, the predictive models do not take account of human action in grazing management. For example, the impacts of grazing have been influenced by a decline in the number of people employed in moorland farming so that there has been a decline of shepherding. Shepherding makes better use of the grazing across the hillslopes and avoids local concentrations which can lead to overgrazing and reduces the need for supplementary grazing (IEEP, 2004).

The impact of reducing or removing grazing in moorlands has been investigated in a number of exclosure studies. Broadly, the changes in vegetation structure and composition are greatest where grazing was previously more intense (Marrs and Welch, 1991). Often it is only when grazing is removed altogether that there are rapid increases in diversity. IEEP (2004) suggested that a combination of different management strategies involving grazing by different animals at different intensities and different times of the year is likely to maximise biodiversity. Grazing systems that maximise the production of just one animal are unlikely to maximise biodiversity.

Test plots on blanket bog at Moor House, North Pennines, excluded since the 1950s, showed that even if grazing is removed heather did not degenerate because of *Sphagnum* layering, which forces heather to generate new shoots as the peat builds up (Adamson and Kahl, 2003). The key findings of the Moor House study were: (1) the impact of removing grazing was least on the lower altitude (550–630 m) deep peat as grazing here was light and at this altitude heather thrives both inside and outside the exclosures; (2) the lower altitude humic soil sites all had less *F. ovina* and *N. stricta* and a greater cover of forbs in the fenced plots compared to the grazed; (3) the higher altitude (690 m) species-poor peat sites showed a dramatic response to fencing. Despite both sites being above the normal altitude for heather it was well established in the fenced plots where there were more species than in the unfenced plots; (4) the high altitude mineral sites (690–830 m) represent some of the most intensively grazed vegetation at the study site. Here *F. ovina* was less abundant in the fenced plot and *Descampsia flexuosa* was more abundant. However, because of the exposed nature of these higher altitude sites, *Carex bigelowii* benefitted from the removal of grazing. Smith et al. (2003) concluded that for blanket peats in northern England, the cessation of grazing would lead to slow structural and species change in vegetation at the mire edge with a loss of bog forming species. Overall, they suggested that, because stocking rates on blanket bogs was generally low, the response of the plant communities would be limited and other external factors such as climate change or atmospheric deposition would be more important in determining bog condition over the next 50 to 100 years.

Grazing is also linked to trampling, which has a significant effect on moorlands. Even vigorous heather cover rapidly dies away on sheep tracks. There have been very few studies of the influence of grazing on moorland hydrology but recent unpublished data from J. Holden and Y. Zhao at the University of Leeds have shown that sheep tracks are important hydrological agents, providing direct connectivity across moorland slopes for water, sediment and pollutants. This is because sheep tracks are compacted and infiltration capacities are reduced so that the infiltration-excess overland flow becomes more common (Table 2). Some summary data on steady-state infiltration rates, hydraulic conductivity, bulk density and proportion of flow moving through macropores are presented in Table 2 for two moorland sites where there are areas with and without grazing. Data were collected using techniques described in Holden and Burt (2003a,c) and Holden et al. (2001). Where grazing occurs, the hydraulic conductivity and infiltration rate is much lower across the hillslope than where grazing has been restricted. It can be seen that just five years without grazing is enough to allow the system to recover towards that of a system that has had no grazing for over 40 years. These changes in hillslope hydrology could be manifested in changes of the river flow and indicate that cessation of grazing may well be a useful tool in reducing flood risk. Sansom (1999) noted that in the north Derwent catchment sheep numbers had doubled between 1944 and 1975 to 24,000 and in that time annual water yield had increased by 25%. It is not known whether other factors contributed to this change but such an increase in grazing is likely to have had some hydrological impact.

Hydrological processes are spatially and topographically controlled. Therefore there may be sensitive parts of a catchment where grazing will have a much greater impact on stream flow (e.g. by compacting valley bottoms) than in other parts of the catchment. The role of sheep tracks also reminds us that if we are to understand the environmental impacts (and make reliable predictions) of reductions in grazing then we need to use spatial modelling techniques that incorporate topographical processes rather than simply rely on lumped models.

3.4. *Afforestation*

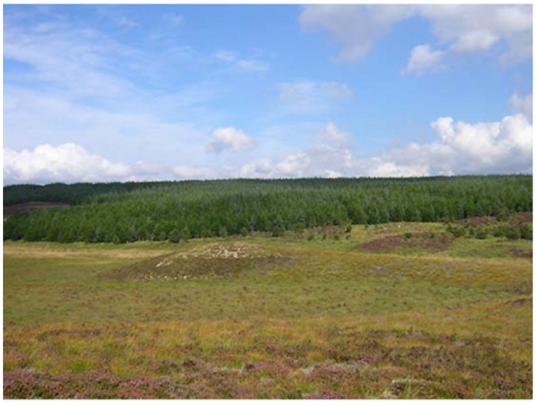
Afforestation has been the main cause of the net loss of moorland habitat over the past century (Fig. 4a). Nine percent of upland UK peatland has been afforested (Cannell et al., 1993) and in Scotland 25% of Caithness and Sutherland peatlands have been affected by afforestation (Ratcliffe and Oswald, 1988). Most of this afforestation has been in the form of commercial coniferous plantations. As the trees grow, the ground cover changes from that of a moorland to that of a forest understorey with sparse *E. vaginatum*, occasional ferns, some mosses, liverworts and lichens. Fertiliser application can decrease *Sphagnum* cover. There is often a tendency for increased nutrient concentrations in the upper half metre of the soil profile, while the

Table 2
Mean summary hydrological characteristics for two moorland sites with and without grazing during 2005 (unpublished data from Y. Zhao and J. Holden)

	Wharfedale				Teesdale			
Grazing condition	None since 1960	None since 2000	2 ha^{-1}	Sheep track	None since 1954	None since 1997	1 ha^{-1}	Sheep track
Mean infiltration rate, mm h^{-1} (n=50)	22.7	20.3	16.3	9.6	22.0	16.3	10.2	5.9
Soil bulk density in upper 5 cm, g cm^{-3} (n=50)	0.033	0.037	0.040	0.088	0.030	0.031	0.037	0.012
Surface saturated hydraulic conductivity $\times 10^{-8}$ cm s^{-1} (n=25)	2753	1388	376	7	8937	4741	931	23
Saturated hydraulic conductivity at 10 cm depth $\times 10^{-8}$ cm s^{-1} (n=25)	11	13	3	1	44	56	13	2
% macropore flow at surface (n=25)	33	35	29	18	36	37	35	21
% macropore flow at 10 cm depth (n=25)	24	32	23	21	29	33	26	21
% occasions overland flow had occurred at 50 locations between bi-weekly visits during 2005	73.3	73.1	80.9	84.1	90.7	89.9	90.5	91.1

n is the number of samples per category.

a)



b)

Fig. 4. Moorland afforestation a) afforestation in Caithness, Scotland, and b) the moorland drains within a mature forest.

application of fertiliser tends to enhance the rate of nutrient cycling by increasing nutrient concentrations in the litter layer (Finer, 1996). Afforestation of moorlands has an impact on biodiversity and is associated with increased earthworms (Makulec, 1991), beetles, moths, plant bugs and slugs with decreases in spiders and wasps (Coulson, 1990). Open ground birds (such as golden plover *Pluvialis apricaria*) are displaced and replaced by forest birds.

Narrowly spaced drainage ditches (ribbon plough furrows) are commonly dug across moorland areas before forests are planted (Fig. 4b). Fertiliser is also often applied. The drains lower the water table and result in associated subsidence of the peat surface due to compression and shrinkage (Anderson et al., 2000). The peat tends to further dry out after canopy closure, and increased interception and transpiration cause a much greater lowering in the water table than drainage alone, further encouraging surface subsidence (Pyatt et al., 1992; Shotbolt et al., 1998) and increasing hydraulic conductivity in the upper layers, often with large scale cracking of the peat. Felling of the trees causes the water table to rise but the water table tends to fluctuate much more than in intact moorlands because of changes to soil structure and enhanced hydraulic conductivity. There have been problems with ditch erosion in the past (Burt et al., 1983) but the implementation of recent guidelines has reduced this problem.

The impact of afforestation of moorlands is not restricted to the planted area alone. Drying and shrinkage of the organic soils can occur at some distance away from the forest depending on local topography and drainage. Bird communities in the surrounding moorland also may be affected up to one kilometre from the forest edge, with reductions in golden plover and dunlin (Moss et al., 1996). Runoff too is affected by afforestation. Streamflow tends to increase in both total and in peakedness with increased low flows in the first years following drainage (perhaps 20 years), followed by decreases in water yield as the forest matures. Water quality may also change downstream as afforested moorland streams often become more acidic with higher concentrations of aluminium. While carbon is taken up by tree biomass as the forest grows, there may be severe depletion of the soil carbon store through enhanced decomposition of the organic soil (Cannell et al., 1993). UK research has indicated that moorland afforestation can result in a net release of carbon dioxide into the atmosphere although the overall effects on greenhouse gases are not yet clear (Holden, 2005b).

Although forest managers now attempt to increase biodiversity through careful planting design (Anderson, 2001), there is a still an increasing area of moorland that is being commercially afforested. There have been major campaigns to protect moorland environments, particularly peat bogs, from additional afforestation (Charman, 2002) and some work is being done on restoring peat bogs once clearfelling has taken place (Anderson, 2001). The costs of such restoration (which usually involves damming of drains and bunding of the area) are very high, however, and restoration has yet to be deemed successful due to changes in soil physical and chemical properties caused by forest furrows and growth. For example, cracks in forested peat soils can form macropore and pipe networks which provide additional drainage and may hamper attempts at moorland restoration after deforestation (Anderson, 2001). Further research is required on methods and feasibility of bog restoration following clearfelling, the influence of bog restoration operations on nutrient cycling and release (e.g. dissolved organic carbon/water colour), and the influence of bog restoration practices on hydrological processes and streamflow.

While there are currently pressures to reduce coniferous afforestation in moorland environments, there are growing demands for increased tree cover in upland catchments and in riparian zones (Gimingham, 1995, 2002). It is likely that there will be increased planting of mixed leaf woodland in the UK uplands. Aims of such planting include: increased water retention in upland soils and aquifers through greater interception, infiltration, reduced storm runoff and groundwater recharge; reduced erosion through greater protection of vulnerable soils/river banks; increased stream water quality through uptake of nutrients in agricultural runoff; increased carbon sequestration associated with increased tree biomass, improvements in landscape aesthetics resulting from greater tree cover and variety; and increased biodiversity.

Much urgent research is needed to develop spatial models for targeting appropriate areas for natural and assisted regeneration of native woodland in upland catchments and river corridors, and minimising any potentially negative impacts (Nisbet and Broadmeadow, 2003). These models need to account for multiple benefits as well as environmental constraints, policy and land ownership issues. Good et al. (2002) have suggested that the potential areas for expansion of woodland, based on an estimation of suitable land for woodland survival, varied from as little as 4% in Northumberland to 34% on Dartmoor. However, when landscape, ecological, agri-economic and archaeological constraints were taken into account, it became apparent that the proportion of land likely to become available for woodland expansion on Dartmoor was actually less then 4%. There are grant schemes for woodland establishment but it is difficult to assess what represents a reasonable balance for woodland expansion over moorlands. Certainly a mosaic of woodland, scrub woodland and moorland will increase biodiversity. However, any change will have both advantages and disadvantages depending firstly on a stakeholder's point of view and secondly on how it impacts different environmental processes throughout the catchment. If a policy to increase semi-natural woodlands is advocated, this will have to occur in parallel with decreased sheep and deer stocking and will need to take account of environmental impacts on, for example, water quality and summer low flows.

4. An uncertain future?

There are likely to be continued efforts to restore UK moorlands through drain and gully blocking and surface revegetation, even though full restoration (to some arbitrary former state) may never be possible in many areas. At the very least, this restoration work aims to reduce sediment loss and reduce water discolouration below that would otherwise have occurred under a warmer climate more prone to drought and under changes in deposition chemistry. There are likely to be reductions in UK moorland burning and grazing, decreases in atmospheric pollution, and some expansion of native woodland into present moorland environments. Each of these changes will have an important socio-economic impact, as well as implications for moorland ecology and hydrology. The changing climate will also impact moorland processes. Nevertheless, change has been a permanent feature of UK moorland environments. Attempting to stop ecological shifts often involves an arbitrary decision which often has no intellectual justification, even though it might seem pragmatic to protect heather moor for the recreational, historical or cultural reasons (Simmons, 2002). Losses of heathland and damage to blanket bogs have been reported for centuries and are not recent phenomenon.

There are a number of more radical changes that we have not fully considered in this review (e.g. land abandonment, transforming the moorlands into arable agricultural zones etc). However, we have chosen to review those responses to the drivers of change that moorland stakeholders felt were more likely in the UK case study. Nevertheless the more radical changes, particularly that of land abandonment should not be discounted and require further research to understand their likely impacts on moorland environments and the wider environment.

This review has indicated that much further research is needed to predict how changes in moorland management will affect the environmental processes. While we have made significant progress, it is evident that a more holistic approach is required. Environmental processes rarely operate in isolation. For example, successful 'restoration' of some moorlands may depend on a combination of factors including changes in burning and grazing regimes and comprehensive drain blocking. Some moorlands may not convert to scrub even if burning is prevented, particularly if grazing is still practiced. If there is a removal of both grazing and burning then moorland may well develop in a new trajectory. Similarly, if heather on blanket bogs is normally prevented from entering the mature (degenerative) phase by peat accumulation, then atmospheric deposition that removes the peat building *Sphagnum* may well have provided conditions promoting mature phase heather. Such a feedback mechanism, if not considered in management planning, may result in the promotion of unnecessary moorland burning when the future decrease in SO_2 deposition will result in such heather control anyway.

Continued maintenance of UK moorlands (e.g. for heather) is important to sporting estates and hill sheep farms as well as for those concerned with the appearance of landscape, amenity and recreation. This includes the tourist industry which benefits considerably from the beauty of open heather moors. However, when the interests of nature conservation are taken into consideration, then there are some who would argue that a much more diverse mosaic is required (Usher and Thompson, 1993; Gimingham, 1995) with the extension of native woodlands and creation of areas of patchy heath and moor with diversity of structure and composition, incorporating elements of scrub and trees. It is the mosaic structure of moorlands that is critical for wildlife conservation. Several scales of mosaic on any one moor are likely to lead to the greatest diversity of vertebrate species. Thus, rather than having a blanket management policy over moorlands, it is necessary to have a spatially distributed policy that allows different areas to be managed in different ways, with different management goals in mind.

This paper has reviewed the science of moorland management and has identified important research gaps. It will be important to understand more about moorland environments in other locations outside the UK and how management has impacted on environmental processes. The work of Buytaert et al. (2005) suggests that there are many similarities and that much of the discussion in this paper is highly transferable to other moorland environments. For example, the páramo moorlands of the Andes are subject to problems of overgrazing, land drainage, afforestation and water quality with similar hydrological and hydrochemical processes (e.g. prevalence of soil piping).

There is also a need to develop predictive models that are spatially explicit in order to take account of spatio-topographic floral, faunal, hydrological, geomorphological and climatological processes. Additionally, social processes need to be incorporated into moorland management models. This approach and the necessary models are now being developed by the authors as part of a new project funded by the UK research councils, DEFRA and SEERAD (see Dougill et al, 2006 and the Sustainable Uplands Project web content, http://www.env.leeds.ac.uk/sustainableuplands). Stakeholder involvement is crucial to such an integrated moorland management approach. It is hoped that such an approach can advance our understanding of the spatial and temporal processes operating in moorland environments. The future for moorlands may be uncertain in many respects but by understanding more about the interlinked processes operating in moorland environments it might, at least, be possible to understand more about the uncertainty.

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