UNIVERSITY OF LIVERPOOL

The role of prescribed burning in moorland management in the Peak District

Thesis submitted in arrordance with the requirements of the University of Liverpool for the degree of Doctor in Philosophy

By

Michael Patrick Kevin Harris

April 2011

Declaration of work carried out

I declare that all the work presented in this thesis, which was conducted at the School of Biological Sciences, University of Liverpool, is my own work, unless otherwise stated. This thesis is submitted in accordance with the requirements of the University of Liverpool for the degree of Doctor of Philosophy.

Work in Chapters 2 (seed banks) and 4 (soils) were carried out with Mr Angus Rosenburgh as part of his Master in Research degree; in both Chapters the data have been substantially re-analyzed and discussed in a wider context.

Signed... Michael Harris

Date....19/4/2011

Abstract

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Michael Harris

The overall aim of this thesis was to develop a better understanding of the role of prescribed burning in moorland management within the Peak District National Park. These moorlands are dominated by heather (*Calluna vulgaris* (L.) Hull). Prescribed burning is a management tool that used routinely to manage moorland vegetation for grouse and sheep production. The aim is to remove the above-ground foliage and allow the *Calluna* to resprout from the burned stems. Normally such prescribed burning is done on a rotational matrix, and the aim is to provide a continual supply of moorland vegetation in different stages. This thesis attempted to answer the following questions:

(a) How degraded are the moorlands in the Peak District, and does prescribed burning affect species density and restoration potential?
(b) What are the environmental factors that influence the response of the plant communities, and how do the constituent species respond after prescribed fire?)
(c) Does prescribed burning affect soil chemical properties?
(d) What factors affect biomass reduction in prescribed fires on upland moorland?
(e) What changes in above-ground biomass, carbon and nitrogen occur during this burning

The context for this work is that British moors are high-priority habitats for conservation and it is increasingly recognized that they provide important ecosystem services (carbon accounting, water provision). A combination of field survey and experiments was used.

A chronosequence study carried out on five replicate moors showed the vegetation was severely depauperate relative to the species that might be expected in pristine moorland vegetation. Moreover, the seed bank was also depauperate and propagules must be added to restore them. There was an increase in species richness immediately following prescribed burning with a subsequent decline with time. Multivariate analysis produced two gradients, a continuum from relatively lichen-rich vegetation to a graminoid-dominated one, and (b) a post-fire growth response of the *Calluna*. *Calluna* was the only species to show increasing growth after burning; all other species were reduced in the oldest vegetation. A similar study of soil properties showed that prescribed burning had a limited effect; some chemical properties changed with the burn-recovery cycle.

In order to develop an improved method of prescribed burning the relationship between fire severity and both fire characteristics and environmental variables was assessed experimentally. The results were inconclusive but suggest that the burns with the highest temperatures were flash fires whereas the burns with the lower residence times were smouldering fires that probably converted more biomass to charcoal). A study of prescribed burns showed that the loss of biomass during prescribed burning was very variable and this almost certainly reflected a range of environmental and management factors. The burning method used in the Peak District is designed to minimize biomass loss and it was demonstrated that in some burns this was very successful. The accumulation of above-ground biomass was measured after burning and the oldest stands had much greater biomass values than literature ones and no sign of an asymptote at 50 years.

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Acknowledgements

This work was funded by personal funds and supported financially by Mr Geoff Eyre, Mr Mark Osborne, The Heather Trust, The Royal Horticultural and Botanical Society of Manchester and the Northern Counties for financial support. The early phase of this work was a pilot that was part–funded by a grant from Moors for Future. Parts of this work were supported under the BiodivERsA FIREMAN research project.

I thank my supervisors, Professors Rob Marrs, Richard Bradshaw and Dr Michael Le Duc, for all their help, support, advice and encouragement. I also thank Dr Hugh McAllister for all his assistance with species identification, Professor Andrew Plater for help with surveying and Mr Angus Rosenburgh and Dr Kath Allen for ongoing scientific help. Mrs Sandra Mather provided many of the graphical outputs, and the National Soil Resources Institute for the soils map of the Peak District.

I sincerely thank all those who helped me in the field and laboratory; Mr Carl Wright and Mr Tony Tollitt who gave up many days to help in sample collection and Mr Les Connor for laboratory support. I would also like to thank all 27 students who gave up weekends to assist me in the field and the administrative staff within the School for the support they have given me throughout my studies. My friends and colleagues past and present I thank them for being there for their friendship, help and support.

I am also grateful to the landowners in the Peak District for allowing me access to their land and their continued support; Geoff Eyre, Mark Osborne, J. Archdale and Neil Taylor. I could not have completed my work without the support of all the gamekeepers and the local community of the Hope Valley in the Peak District.

On a personal note, I would like to thank my parents, my wife Chuthamat and all my friends. They have been a constant source of support throughout my research.

Chapter 1

Heathland and moorland ecosystems: an introduction to their history and current management

1. Introduction

The aim of this thesis is to provide a better understanding of prescribed burning for moorland management in the UK, with particular reference to moorlands in the Peak District of Derbyshire. Moorlands are important habitats within the UK because they provide a range of ecosystem services (Millennium Ecosystem Assessment, 2005), provisioning (water supplies), regulating (carbon sequestration) and cultural (conservation, recreation, aesthetics).

The term moorland is local British one, which is often used interchangeably with heathland. Traditionally, these terms were used to denote landscapes either at different elevations, heathland less than 250 m and moorland greater than 250 m (Webb, 1986), or at different degrees of waterlogging, heathland being drier and moorlands wetter (Gimingham, 1996). In a large review of British communities Rodwell (1991, 1993) based his major descriptions on the plant community types that occur in heathlands and moorlands, i.e. Mires, Heaths and Upland grasslands. Here, Gimingham's (1972) definition of heathland will be used. This was based on earlier work of Warming (1909) and is "a treeless tract that is mainly occupied by evergreen, slow-growing shrubs which are largely Ericaceae". In the UK, the main dwarf-shrub species is usually *Calluna vulgaris* (referred to as *Calluna* hereafter in Chapter 1); otherwise nomenclature follows Stace (1997) for higher plant species and Atherton *et al.*, (2010) for bryophytes. On upland moorland, the definition should also be expanded to include communities with a larger component of graminoids (grasses, sedges and rushes) and bryophytes, especially *Sphagnum* spp.

To provide a back ground to the work described in this thesis, it is essential to understand: (1) heathland and moorland ecosystems in world context, (2) how moorlands developed historically in the UK, (3) the moorland communities at the present time along, their conservation status, (4) current threats to moorland, (5) fire ecology and its relevance for moorlands in the UK, and (6) the Peak District as a regional moorland area within the UK.

1.1. Heathlands and moorland ecosystems in a global context

Whilst shrub-dominated communities occur in many parts of the world, for example the Macquis of Mediterranean areas, the Fynbos of South Africa and the Cerrado of South America, the type locale of heathlands and moorlands occur along the Atlantic coasts of northern Europe, with some examples extending as Far East as Poland (Gimingham, 1972, 1996). All of these shrub-dominated communities tend to be found on infertile soils where some factor or factor(s) prevents succession to woodland. In the Macquis, Fynbos and Cerrado naturally-occurring fires are important in preventing tree colonization (Sugihara *et al.* 2006; Thomas & McAlpine, 2010).

In the UK, the natural vegetation over most of the country without human influence would be a woodland, a mixed deciduous forest over most of England and southern and central Scotland, and a northern, boreal, taiga-type, forest in northern Scotland (Rackham, 1980, 1986). *Pinus* and *Betula* arrived immediately after the ice retreat with *Alnus* and *Ulmus* also present in detectable quantities across most of the Boreal region. Turner and Hodgson (1979) showed that the Pennine area of northern England was covered by broad-leaved rather than *Pinus* forest 5-7,000 BP. The current heathland and moorland communities would have been present in clearings within the wood, created naturally through gap creation after tree death. In the more northerly areas, the Scottish native pinewoods provide an insight into the type of moor-wood mosaic and dynamics that might occur with little influence of man. Here, the woodlands have a relatively light tree density and through time have been postulated to move across the moorland understory (Stephen & Carlisle, 1959). At the present time the large red deer (*Cervus elaphus* L.) herds present in the Highlands preclude tree regeneration but this is because the deer have no large carnivores to control their numbers (Baines *et al.*, 1994).

In the UK, therefore, heathlands and moorlands are usually sub-seral, or plagio-climax communities, extending from sea level to the sub-montane zone (Gimingham, 1972; Rodwell, 1991). Usually there is a successional trend in most heaths and moors towards woodland that is prevented my man's activities (Gimingham, 1972, 1996; Webb, 1986). There are a few areas where the heath/moor ecosystem might be the final "climax" community, for example at the high altitude fringe of the forest tree line and in very exposed or waterlogged places, but in most places heathlands and moorlands will, if left unmanaged, be colonized by trees and develop into woodland. This process is most evident in lowland British heaths where the conservation objective on many sites is to prevent tree and/or bracken invasion (Marrs, 1984, 1985, 1988).

Many upland moors in the UK are on peaty soils. Rydin and Jeglum (2006) reproduced a useful schema from Ruuhijärvi (1983), which separated different types of peat habitats in Finland on the basis of gradients of moisture and mineral nutrition. Most of the Fenno-Scandinavian and Russian peatland fall into variants of this schema (Rydin & Jeglum, 2006). The moorlands of the UK, many of which are dominated mainly by dwarf shrub and *Eriophorum* spp. (Rodwell 1991, 1993), fit into the drier, oligotrophic part of this schema (dwarf-shrub pine bog –*Eriophorum* pine bog). The substrate pH of most bogs at northern latitudes is between 3.5-4.5 (Rydin & Jeglum (2006) because of the acidifying effect of the dominating *Sphagnum* species and the low buffering capacity of the rainfall. This is consistent with the acidic nature of most upland heaths in the UK (pH 3.5-4.5 over a range of vegetation types on peat in the north Pennines (Marrs *et al.*, 1986).

1.2. The historical development of heathlands and moorlands

Historical evidence from both pollen diagrams and the occurrence of that "bog timber", trees buried and preserved by peat provides evidence that forests usually preceded heathland/moorland (Faegri, 1940; Conway, 1947; Gimingham, 1972; Godwin & Tallantire, 1951; Tallis, 1964a, 1964b). This tree decline is summarised in Fig. 1.1, where the increase in heath species (Ericales) starts ca. 5,500 BP. How then did these heathlands and moorlands develop? Essentially, this has come about through a combination of effects, both climatic changes and the influence of man.) The shift from forest to heath would have been a gradual process; it was first detected as long ago as the Atlantic period (ca. 6000 BP), although this process increased through the Bronze age (ca. 3000 BP) and up to the beginning of the Iron age (ca. 2500 BP) (Pennington, 1970). The transition from forest to that moor coincided with climatic changes, when conditions became cooler and more oceanic, but human influence was also important in this change as trees were removed for timber and to create land suitable for grazing (Pennington, 1970).

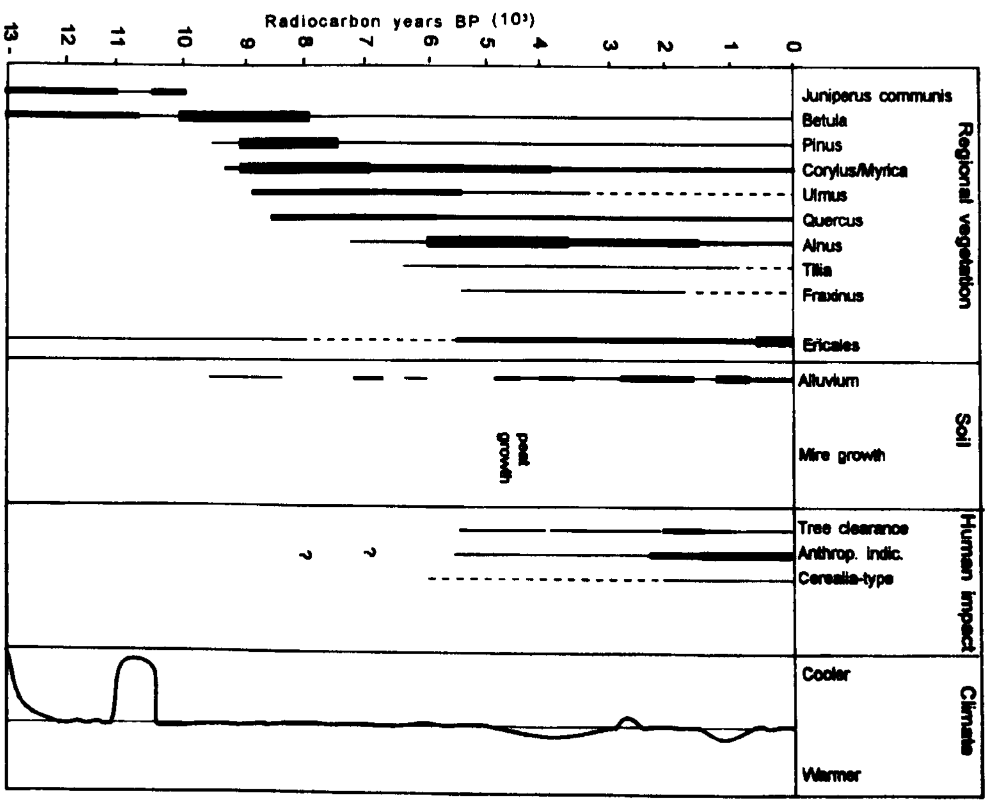

Fig. 1.1. Events stratigraphy for upland England (Greig, 1996).

For lowland heaths in the UK, Rackham (1986) suggested that large-scale creation of lowland heath started in the Neolithic (6,000-7,000 BP) and increased through the Bronze Age (3,500-4,500 BP) and suggested, "there was nothing to support the common view that heaths were lightly forested". This implies that lowland heaths were created and maintained mainly by human activity; they would have been used for some light agriculture and grazing. He also argued that burning would have been practiced much less that in upland situations. Since then lowland heathlands have been extensively managed using combinations of burning, cutting, grazing and turf cutting, with several cultural variants across Europe (Kaland, 2000). An example of this is plaggen in the Netherlands where (1) turfs were removed each year and taken to a holding barn, (2) sheep (*Ovis aries* L.) were shepherded daily into different parts of the heath and returned to the barn at night where their urine and faeces were added to the turfs, and (3) the resultant mixture was added to in-bye land to improve soil fertility for crops (Karland, 2000).

In upland moorlands in the UK the situation is more complex than for lowland heaths because of the peat development (Rackham, 1986). The accumulation of peat has continued at varying rates since the Boreal-Atlantic transition period (ca. 7,500 BP) when rainfall increased markedly (Pennington, 1970) with *Sphagnum* spp., *Eriphiorum* spp., and *Calluna* as peat-forming species (Heal & Smith, 1978). According to Rackham (1986), moorland was not explicitly identified in the Domesday Book rather they were merely gaps between settlements, although it does identify how the land was parcelled up between the King's supporters. Rackham then goes on to state that pressure on land in the twelfth and thirteenth centuries pushed cultivation from lowland areas up into the moors. This culminated in large flocks of sheep being introduced into some upland areas of England, for example by the Cistercian monks, but he argues that the flock sizes were insufficient to change whole landscapes. There is also evidence to suggest they introduced Red deer (*Cervus elaphus*). From the late 1700s there was change in land management practices in upland Britain brought about by socio-economic change in human society. Large parts of the uplands were transformed into large estates and much of the moorland given over solely to large-scale sheep grazing (Darling & Boyd, 1969) and sporting interests (red grouse, *Lagopus lagopus scoticus* Latham). This type of ownership structure has persisted to the present day with the one exception, the ownership and management of some estates has been taken over by charitable bodies (National Trust, National Trust for Scotland, John Muir Trust, RSPB and others) where the management aim is more sympathetic to the conservation of landscape and nature.

To summarize, most of the heathlands and moorlands that exist today in the UK are cultural landscapes (Thompson *et al.*, 1995), i.e. they are plagio-climax communities, or arrested successions (Gimingham, 1972, 1992), created by human activity and they require management to maintain them as heathland. As most heathlands and moorlands are found on infertile soils (usually podzols or peats, Gimingham, 1972), the management acts to maintain low nutrient supplies through the continued removal of nutrients combinations of managed grazing, or through the removal of turves or vegetation (Marrs, 1993; Kaland, 2000).

1.3. Moorland communities at the present time and their conservation status

On the world scale the heathlands and moorlands are a relatively small vegetation type, As such they are protected under both European and UK legislations, for example at EU level, Special Protection Areas (SPAs), Special Areas of Conservation (SACs) and Natura 2000, and at UK level, National Nature Reserves and Sites of Special Scientific Interest (SSSIs). There are Biodiversity Action Plans for lowland heath, moorlands, blanket bog and many constituent species. In the uplands of the UK large tracts are notified as SsSSI/AsSSI for their moorland communities, including at least 42,000 ha in England, 34,000 ha in Wales, 7,000 ha in Northern Ireland and 152,000 ha in Scotland (BAP, 1999). Of the moorland managed as grouse moors 49% are designated as SsPAs and SsACs, the former for the rare birds they support and the latter for their plant species and plant communities (Table 1.1). Many upland moors are managed for grouse shooting and of these 66% are protected as Sites of Special Scientific Interest and 45% of grouse moors carry all three designations (Countryside Alliance, 2010) - making them one of the country's most important habitat types from a conservation viewpoint.

Substantial areas of heathland and moorlands lie in countryside designated as National Park, National Scenic Area (NSA) or Area of Outstanding Natural Beauty (AONB), and as such they are protected for their aesthetics and landscape value as well as their conservation interest. Within National Parks they are subject to additional planning controls.

Major changes in moorland vegetation have been observed over the 20th century. Approximately 25% of heather moorland vegetations as been estimated to have been replaced by grassland (*Molinia caerulea, Nardus stricta*), *Eriophorum* spp., *Pteridium aquilinum* or coniferous plantations since the 1940s in both England and Scotland (Moorland Working Group, 1998; Tallis, 1981). For the Peak District and Cumbria, 36% of moorland has been replaced since the early part of the 20th Century (Anderson & Yalden, 1981; Felton & Marsden, 1990). The general trend appears to be toward more productive and less diverse vegetation dominated by species with a lower agricultural value. This transition has been widely blamed on overstocking of sheep and the increase in deer numbers brought about by lack of top predators, although other factors such as nitrogen deposition have also been implicated.

Much of the remaining moorland has been judged to be in unfavourable condition by the statutory authorities (BAP, 2010), i.e. unfavourable, unfavourable recovering and unfavourable declining. The reason for this poor state is a combination of over-grazing and trampling (mainly by sheep), trampling pressure on sites where there is a large visitor pressure, which has led to severe erosion problems, pollutant impacts (SO_2, NOx) derived mainly from atmospheric deposition, inappropriate burning and potentially climate change (Holden *et al.*, 2006; McDonald *et al.*, 2000). Blanket peats, for example, have suffered severe degradation in many parts of the UK and particularly in the English Pennines (Anderson & Yalden, 1981). Erosion in the southern Pennines has been very severe over the past 200 years, deep gulleys are now present where there is no vegetation and through which there can be substantive peat loss (e.g. Bower, 1961; Tallis, 1965; Labadz *et al.*, 1991; Yeloff *et al.*, 2005).

Recently, the focus on peat conservation has been intensified because of the realization of the importance of carbon storage and cycling in mitigation of global warming (IGPCC, 1990; MEA, 2005). The uplands of the United Kingdom have a large carbon store in peat, and this needs to be conserved. Britain's peatlands (most of which are located in uplands) store around 3 billion tonnes of carbon – more than all the forests of France and UK combined (Worrall *et al.*, 2003). The planned policy is to develop aggrading peatlands that sequester carbon, and limit peat degradation (Clay *et al.*, 2010). If this is achieved then the peat carbon stock will count towards carbon credits under the Kyoto Protocol (Watson, 2001); Grace, 2004). Billet *et al.* (2010) reviewed the current state of knowledge and concluded that some UK peatlands act as carbon sinks although the size of the sink has declined over the last 100 years. Historically, carbon accumulation rates were in the range of -35 to -209 C m^{-2} yr^{-1} (-ve flux = uptake from atmosphere). However, more recent data showed two sites (Auchencorth Moss, SE Scotland; Moor House, N England) with lower accumulation rates of -56 to -72 g C m^{-2} yr^{-1} with large inter-annual variation.

1.3.1. Communities of conservation importance on moorland

The plant communities of the UK have recently been described using a phytosociological framework, the National Vegetation Classification (NVC, Rodwell 1990, 1991, 1992, 1995, 2000). The NVC classes represent idealized communities based on multivariate classification of a very large number of relevés covering the spread of variation in UK vegetation with a high conservation value. Many of the communities described represent semi-natural ones, including mires, heathlands and grasslands. The relevés were taken from a combination of legacy datasets sampled between 1950 and 1970 and new data collected between 1974 and 1990. Whilst, the NVC represents a snapshot of British plant communities in time, it is a very useful framework to assess conservation status, essentially providing reference communities against which other studies can be related.

On British upland moors 19 plant communities have been described; this is 23% of total GB complement of upland communities and five are virtually confined to the UK and a further six are better represented in the UK than elsewhere (Table 1.1). Of these, 13% are listed under the EC Habitats Directive "92/43/EEC". There are 46 species of birds that occur regularly on heather moorland for feeding and/or breeding in England and Wales (Table 1.2).

Table 1.1. Important heather and associated moorland communities in the UK under the International and EC Habitats Directive (1994).

NVC Code	Community Name	International Importance/ EC Habitats Directive	No Plant Species	
			Mean (range)	Total
H12	*Calluna vulgaris-Vaccinium myrtillus*	I/EC	17	4-24
M19	*Calluna vulgaris-Eriophorum vaginatum* blanket mire	I/EC	19	7-33
H10	*Calluna vulgaris-Erica cinerea* heath	I/EC	20	5-58
H9	*Calluna vulgaris-Deschampsia flexuosa* heath	I/EC	8	2-15
H8	*Calluna vulgaris-Ulex gallii*	UK/EC	13	4-32
H4	*Ulex gallii-Agrotis curtisii*	UK/EC	11	5-19
H16	*Calluna vulgaris-Arctostaphylos uva-ursi* heath	-/EC	19	8-31
H21	*Calluna vulgaris-V.myrtillus-Sphagnum* heath	UK/EC	29	10-46
M18	*Erica tetralix-Sphagnum papillosum* blanket mire	I/EC	17	8-30
M17	*Scirpus cespitosus- E. vaginatum* blanket mire	UK/EC	20	8-38
M16	*E.tetralix-Sphagnum compactum* wet heath	UK/EC	16	2-28
M15	*S. cespitosus-E.tetralix* wet heath	I/EC	18	6-57
M20	*E. vaginatum* blanket/raised mire	I/EC	11	5-20
U2	*Deschampsia flexuosa* grassland	-/-	9	3-16
U4	*Festuca ovina-A. capillaris-G.saxatile* grassland	-/-	22	7-62
U5	*Nardus stricta-G.saxatile* grassland	-/-	21	6-42
U6	*Juncus squarrosus-F.ovina* grassland	UK/-	15	7-36

Plant community codes according to the National Vegetation Classification (Rodwell, 1991, 1992). International importance codes taken from Thompson & Sydes (1995). There are three categories of communities: (a) Those that have no, or rare, close affinities outside the UK and Ireland (=UK); (b) those that are very localized globally, but especially well developed in the UK (=I); and (c) those listed under EC Directive 92/43/EEC on the Conservation of National Habitats and Wild Fauna and Flora (note that species- rich communities of U4 and U5 are listed under this Directive). Number of plant species taken from Rodwell (1991, 1992). Numbers are as counted in 2 or 4 m^2 quadrats. The rare species are those occurring in <100 10km grid squares in GB. Another two communities (*Sesleria albicans-Galium sterneri* grassland and *Festuca ovina-Agrostis capillaris-Thymus polytrichus* grassland) are locally extensive in Carboniferous limestone in the uplands.

Table 1.2. Birds identified listed under the Biodiversity Action Plan that either breed on upland moorland, or use moorland as an important part of their habitat (BAP, 2009)

Birds Virtually Confined to Heather Moorland	Birds that Breed mainly on Heather Moorland	Moorland Provides Major Breeding Habitat	Moorland Provides Locally Important Breeding Habitat	Moorland Provides Important Feeding Habitat
Red Grouse *Lagopus lagopus scoticus*	Golden Plover *Pluvialis apricaria*	Greenshank *Tringa nebularia*	Twite *Acanthis flavirostris*	Golden Eagle *Aquila chrysaetos*
	Merlin *Falco columbarius*	Curlew *Numenius arquata*	Wren *Troglodytes troglodytes*	Peregrine *Falco peregrinus*
	Hen Harrier *Circus cyaneus*	Short-eared Owl *Asio flammeus*	Wheatear *Oenanthe oenanthe*	Raven *Crovus corax*
		Meadow Pipit *Anthus pratensis*	Lapwing *Vanellus vanellus*	Buzzard *Buteo buteo*
		Whinchat *Saxicola rubetra*	Common Snipe *Gallinago gallinago*	Kestrel *Falco tinnunculus*
		Teal *Anas crecca*	Redshank *Tringa totanus*	Red Kite *Milvus milvus*
		Black Grouse *Tetrao tetrix*	Black Headed Gull *Larus ridibundus*	Common/hooded Crow *Corvus corone cornix*
		Dunlin *Calidris alpina*	Oystercatcher *Haematopus ostralegus*	Dotterel *Charadrius morinellus*
		Common Gull *Larus canus*	Whitethroat *Sylvia communis*	Reed Bunting *Emberiza schoeniclus*
		Stonechat *Saxicola torquata*	Pochard *Aythya ferina*	Osprey *Pandion haliaetus*
		Skylark *Alauda arvensis*	Pheasant *Phasianus colchicus*	Goshawk *Accipiter gentilis*
		Ring Ouzel *Turdus torquata*	Mallard *Anas platyrhynchos*	
		Grey Partridge *Perdix perdix*	Willow Warbler *Phylloscopus trochilus*	
		Fieldfare *Turdus pilaris*		
		Wigeon *Anas penelope*		
		Nightjar *Caprimulgus europaeus*		
		Whimbrel *Numenius phaeopus*		

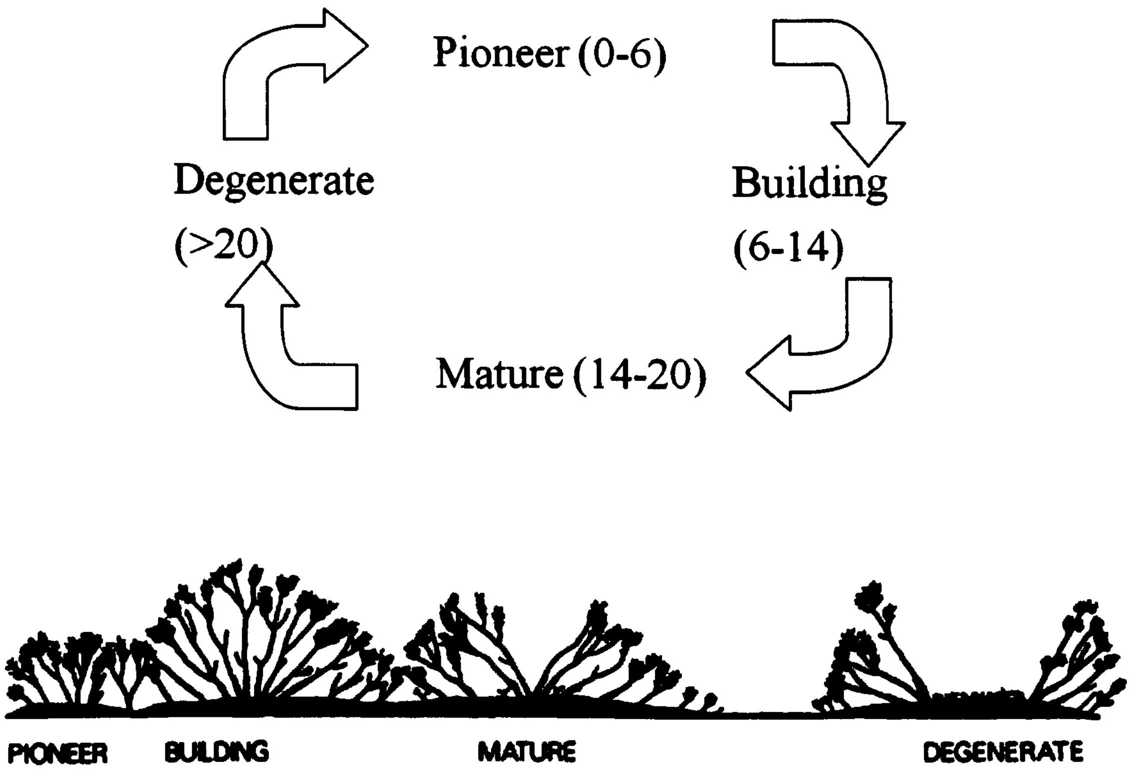

Fig. 1.2. The Watt four-phase life-cycle of *Calluna*: (a) the basic cycle for moorland with approximate age ranges in years; (b) diagrammatic profile after Watt (1947, 1955).

1.3.2. The *Calluna* life-cycle

Calluna dominates many moorland communities, and moorland management has to take account of the *Calluna* life-cycle. Watt (1947) first described a four-phase *Calluna* life-cycle (Fig. 1.1). The plants start in the pioneer phase, develop through building and mature phases, and eventually degenerate, when the cycle in its simplest form repeats. Burning and cutting interrupts this cycle, and if burned or cut shoots remain then resprouting occurs and the pioneer phase re-established quickly from the burned/cut shoots (Miller & Miles, 1970). Usually burning is carried out in the mid-late-building phase when the plant has the greatest density of stems capable of producing new sprouts, and the ability to resprout declines with age (Miller & Miles, 1970). Watt's original model did not include layering, where stems become buried and root vegetatively in the underlying litter, and this process has been demonstrated in many moors with a wet oceanic climate (McDonald *et al.*, 1995). Here the mature and degenerate plants can establish new pioneer plants by layering (Marrs, 1988). Layering is particularly prevalent in wet areas with a deep bryophyte layer, and in certain peat soils the *Calluna* may be able to continue growing vertically, producing new roots in the litter layer, and may not reach its degenerate phase (McDonald *et al.*, 1995).

1.4. Current impacts on heather moorland

1.4.1. Atmospheric pollution

There have been significant changes to atmospheric deposition chemistry across the UK over the past 250 years since the start of the industrial revolution and moorland areas near industrial centres have been most affected. The major atmospheric pollutants are acidifiers (sulphur dioxide (SO_2), nitrogen oxides (NO_x), although many moorland areas are also contaminated by lead from mining spoil (Longhurst, 1995). In the second half of the twentieth century new legislation has helped reduce emissions (Quart, 1954). Historically, the major acidifying pollutant has been SO_2 emitted from fossil fuel combustion (Strode *et al.,* 2009), but this has been augmented more recently with NOx. Acid deposition, which reached a peak in the early 1980s, has had a wide range of impacts upon soil and vegetation. Although global emissions of N are expected to continue to increase (Galloway *et al.*, 2004) N emissions in many European countries, including the UK, have been in decline since 1990 (NEGTAP 2001, Defra 2008). As little research to date has focused on the response of N polluted ecosystems to reductions in N deposition, and the mechanisms of recovery are relatively unknown, there is now a growing need to determine how N polluted systems may recover following reduced N deposition.

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, base-poor soils makes these moorland ecosystems particularly vulnerable to acidic deposition, and has been linked to major changes in species composition in moorland environments (Lee *et al.*, 1993) and widespread loss of bryophyte and lichen species (NEGTAP, 2001). Where acidic deposition has been very high, such as in the southern Pennines, *Sphagnum* has been almost eliminated. Field experiments with *Sphagnum* have shown that it quickly succumbs to the application of SO_2 in solution (Ferguson & 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.

Over the long-term, acidic deposition leads to a decline in soil acidity as it increases the leaching of base cations (Ca and Mg) and increases the number of exchange sites in soils occupied by aluminium species and hydrogen ions. A decline in soil pH also increases the solubility in the soil of metals, such as aluminium, manganese, lead, cadmium and zinc, which can be toxic to plants (NEGTAP, 2001). This can lead to decreased plant growth, changes in plant communities, reduced litter decomposition rates, increased surface accumulation of litter (Sanger *et al.*, 1994) and delayed nutrient cycling. Soil acidification gradually leads to acidification of waters draining from such soils (Cresser & Edwards, 1987), and reduction in diversity of river invertebrate communities (Weatherly *et al.*, 1987, 1990).

A great deal of pollutant damage to vegetation has been ascribed to increases in acid rain (mainly SO_2) since the start of the industrial revolution, which accelerated after World War II. Over the last two decades, sulphur deposition in the UK has declined by 60% as a result of measured taken to reduce emissions (Fowler *et al.*, 2005). 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 vegetation and soils respond. While sulphur deposition has declined rapidly, atmospheric deposition of reactive N compounds has increased mainly from vehicle emissions, and has now reached inputs of 40 kg N ha^{-1} yr^{-1} over large areas of the UK (NEGTAP, 2001). The increased N deposition onto moorlands represents a particular threat because the plant species are adapted to very low levels of N and occur on soils of low nitrogen content.

1.4.2. Drainage or moor gripping

All moorland areas in the uplands serve catchments for domestic and industrial water, an important ecosystem service (MEA, 2005). The water that runs off the moors is often conserved in reservoirs and after treatment is passed to the customer by utility companies. These companies require a large quantity of high-quality water, free of colour, which has to be removed at large cost. Colour is due to dissolved organic compounds (DOC) that may consume free residual chlorine used in purification and can form trihalomethanes that cause unpleasant tastes. Law in the UK limits the concentrations of these chemicals as they are also potential carcinogens, (Hsu *et al.*, 2001). Moorland management can clearly impinge on both the water quantity and quality leaving the catchment.

Drainage of peatlands by open channels is common in many European countries. In the UK about 1.5 million ha of the country's 2.9 million ha of peat has been drained (Milne & Brown, 1997), a large fraction in the latter half of the 20th century, with a peak in the 1970s, when agricultural subsidies (70% of cost) were in place to promote moor gripping, i.e. the installation of drains cut into peat (Stewart & Lance, 1983). The aim was to drain the moorland and improve the vegetation productivity for sheep (Stewart & Lance, 1991) with grips spaced usually 20 m apart parallel to slope contours. The grips feed into streams or drains that run down slope. The drains are trapezoidal in section, typically 50 cm deep, 90 cm broad at the top and 40 cm broad at the base.

Stewart and Lance (1983, 1991) assessed the effect of such moor gripping on a variety of ecosystem variables, and in spite of their widespread use there was no documented evidence that they produced any of the supposed benefits. There was a slight effect on the water table, confined to a few metres either side of the ditch. The effect of draining on the vegetation was low, with a shift in cover abundance, a reduction in species typical of wet habitats and an increase in those with less demanding water requirements. There was very little evidence that drainage was successful in improving plant productivity. Instead peat drainage has been associated with environmental degradation (e.g. increased flood risk) and increased particulate carbon release both from drain channels and consequent soil pipe development in desiccated peat (Holden, 2006; Holden *et al.*, 2004). Mitchell & McDonald (1995) showed that areas of upland peat where there were more ditches were greater sources of water colour.

Conservation agencies and other NGOs are now blocking peat drains to try to improve peatland habitat with peat dams, plastic pile dams, heather bales, and wooden dams (Holden *et al.*, 2006). If such grip-blocking can also be shown to have a positive impact on water quality, and allow plant communities typical of wet bog habitats to develop, they are likely to be used more frequently in the future. However, grip- blocking is expensive. Blocking ditches using plastic dams can cost over £30,000 km^{-1}, whereas blocking drains using peat dams is substantially cheaper at £7,000 km^{-1}. Wallage *et al.* (2006) compared DOC concentrations in peat soil water in an un-blocked catchment and a catchment that had been blocked for 4 years in Upper Wharfedale, Yorkshire. DOC concentrations were lower in the blocked catchment suggesting that the blocking had been successful.

Large increases in DOC concentration have been reported in Northern Europe (Skjelkvale *et al.*, 2001; Freeman *et al.*, 2001a; Worrall *et al.*, 2004b), North America (Driscoll *et al.*, 2003; Stoddard *et al.*, 2003) and in Central Europe (Hejzlar *et al.*, 2003). Often, high DOC and its associated water discolouration are prevalent in catchments with an extensive peat cover (Aitkenhead *et al.*, 1999). Peat is an important store of terrestrial carbon (Holden 2005), and as such also provides an important ecosystem service (MEA, 2005). It is not known why there has been a recent increase in DOC in peat catchments but several possibilities have been suggested, including: increasing air temperature (Freeman *et al.*, 2001a); changing land management (Worrall *et al.*, 2003); increasing or decreasing pH (Krug & Frink, 1983; Bouchard, 1997; Grieve 1990a,b; Kullberg & Petersen, 1987); changing water flow patterns (Tranvik & Jansson, 2002); eutrophication (Harriman *et al.*, 1998); increasing atmospheric CO_2 concentrations (Freeman *et al.*, 2004); severe summer droughts (Freeman *et al.*, 2001; Worrall *et al.*, 2004 a) and decreasing acid deposition (Evans *et al.*, 2005); Clark *et al.*, 2005). Clearly, water companies cannot control climate or atmospheric deposition, so any influence they have must be brought about by a change in land management.

The release of carbon from peat is often considered to be largely controlled by water table depth (Holden, 2005); it is limited in anaerobic conditions and promoted in aerobic conditions which allow increased soil CO_2 respiration and peat decomposition (Jones & Mulholland, 1998) (Glenn *et al.*, 1993); Funk *et al.*, 1994); Bubier *et al.*, 2003). Higher water tables reduce oxygen concentrations, but there is increased production and decreased oxidation of CH_4 (Huttunen *et al.*, 2003). Worrall & Burt (2005) have successfully modeled DOC flux from a peat-covered catchment where DOC production within an aerobic zone was controlled by the depth of the water table. Therefore, raising the water table is a possible mechanism for restricting DOC production, and hence water discolouration in peat catchments.

Research on water table, peat, carbon, water and greenhouse gasses has, so far, produced conflicting results about the relationship between moorland management, climate change and water quality (Holden *et al.*, 2006, Holden *et al.*, 2007, Orr *et al.*, 2008). It is not yet clear what effects management may have, in either contributing to, or the mitigation of, the release of green house gases or on the quality of drinking water (Worrall & Burt, 2004; Evans *et al.*, 2006). For instance, blocking moorland drains and re-wetting peat can lead to an increase in methane emissions, the global warming potential of which is 24.5 times higher than that of carbon dioxide (Reed *et al.*, 2009 b). Burning (prescribed or wildfire) will almost certainly lead to an increase in DOC in run-off as a result of loss of biotic control (*sensu* Bormann & Likens, 1978).

1.4.3. Sheep

Sheep grazing has played a major part in the management of most moorland areas over the last 200 years (Gimingham, 1972). Sheep numbers increased steadily as a result of agricultural production policies brought in, first after the Second World War, and secondly as a result of the EU Common Agricultural Policy (Ross *et al.*, 2003). In the UK, until 2000, sheep subsidies to farmers were paid on a headage basis through the Hill Livestock Compensatory Allowance (HLCA), which encouraged high sheep numbers. More recently this system has been changed, partly to reduce sheep numbers for environmental reasons, and farmers are now given a Single Farm Payment based on area and nature of the land (Lowe *et al.*, 2002).

The impact of sheep grazing on the species composition of moorlands was first demonstrated by Jones (1967) who showed that increased sheep densities changed *Calluna* to a grass-dominated sward, with a reversal if the sheep were removed. The impacts of high sheep grazing intensities, coupled with excessive burning frequencies were suggested as a major cause of forcing *Calluna*-dominated moorland to a grass dominated one (Miles, 1987). It is important to assess the distribution of the sheep across the moorland; absolute densities are of very limited value. The sheep need careful shepherding to prevent large numbers from concentrating for too long on any one part of the moor; if this is not done then there will be considerable damage to the *Calluna,* and grass will take over (Grant *et al.*, 1981); Palmer *et al.*, 2003). Farmers also often need to provide supplementary feed for sheep, especially in winter (Van den Berge *et al.*, 2009). Where this is required, the use of permanent fixed feeders is to be avoided as they will encourage large numbers of sheep at one spot and there will be considerable vegetation damage (Van den Berge *et al.*, 2009).

There is also an important interaction between sheep and red grouse. Sheep tend to reduce the *Calluna* height and keep it free from snow in winter. The grouse are often found feeding where the sheep have trampled down the snow (Dayton, 2006).

1.4.4 Sporting interests

Grouse shooting (red grouse) is a major income generator in upland communities. For example, in Scotland in the early 1990s, grouse shooting provided £14.7 million in revenue and supported 904 full-time jobs in the hotel industry (McGilvray, 1995). The larger sporting shooting industry has been calculated to be worth £1.6 billion to the UK economy, with 13.33% or £120 million, spent on grouse-shooting in good grouse years (PACEC, 2006).

Red grouse is a territorial bird and is usually monogamous. The cocks defend their territories over winter, and as the ground in many areas is fully occupied, the territory size is the inverse of cock breeding density (Moss *et al.*, 1995). They are found mainly on open moorland where its main food plant (*Calluna*) is the dominant plant species; they eat its short, succulent shoots and they require a mixed vegetation structure to accommodate feeding and shelter. Nesting takes place in a scrape on the ground amongst the cover of older *Calluna.* Between 6 and 14 eggs are laid in April or May. The chicks are feathered when they hatch and within a few hours are able to leave the nest. To begin with, the young feed on insects that are most abundant near the wet and boggy flushes on the moors then they switch to feed on *Calluna* (Coulson. 1994).

Populations of red grouse cycle with a periodicity of ca 10-11 years in north-east Scotland (Moss *et al.*, 1996). There has been a great debate on the causes of these cycles including (a) age-structural population dynamics and associated behaviour, (b) maternal nutrition, (c) genetic behaviour, (d) host-parasite effects (caecal threadworm *Trichostrongylus tenius*) and (e) predator-prey relationships. The age-structural population dynamics and associated behaviour theory was favoured by Moss *et al.* (1996).

Like other ground nesting birds, grouse, their eggs and chicks, are very vulnerable. They suffer badly from predation and disease (worms, *Trichostrongylus tenius* and a virus, louping ill). Where predation and disease are reduced bird numbers increase (Game & Wildlife Conservation Trust, 2010).

1.5. Fire ecology and its relevance for moorlands in the UK

Fire is a factor that affects many ecosystems both naturally and as a means for man to manage land. In the creation of moorland fire was almost certainly used as evidenced from charcoal remains in peat (Pennington, 1970). Fire can have both positive and negative benefits on ecosystems, positive in that it creates a disturbance and this allows community regeneration (Miles, 1971) and negative in that very severe damage results in the loss of the entire organic matter in the system. In some parts of the world fire natural fires are commonplace (Sugihara *et al.* 2006; Thomas & McAlpine, 2010); this is especially true in sub-tropical savanna-scrub systems and Mediterranean Macquis scrub ecosystems. Fire has been used by humans from a very early time for cooking and to help drive game and clear land for settlement, grazing and farming (Rackham, 1980). Since medieval times, burning has been carried out to improve grazing for sheep (*Ovis aries)* and red deer, a practice that practice continues in some places today (Rackham, 1986). However, since the middle of the nineteenth century prescribed burning has become common practice in upland Britain for the management of sheep and red grouse (Richards, 2004).

However, in all heathlands and moorlands there is the constant risk of wildfire. The risk is particularly high in dry years, in areas subject to high tourist pressure and near the urban-moorland fringe (Peak District National Park Authority, 2006). When this occurs there may be very severe impacts on the moorland ecosystems with large losses of vegetation and peat (Maltby *et al.*, 1990). There is an increasing realisation that wildfire risk might be able to be manipulated through the use of prescribed burning.

1.5.1. Fire ecology

(a) Fire impact terminology

Keeley (2009) has recently reviewed the terminology associated with fires and suggests the following terminology for describing studies on fire ecology:

1. Fire intensity – defined as the energy released
2. Fire severity – defined as the loss of organic matter
3. Ecosystem response – defined as the recovery of the ecosystem and includes processes like soil erosion and vegetation recovery

4. Societal impacts – loss of life or property, suppression costs.

These descriptions illustrate the wide-ranging scope of studies on fire. In this thesis, this terminology will be used throughout, but the focus here is mainly on fire intensity, fire severity and ecosystem response. This does not mean that there is no societal impact, but it is not part of the research focus in this thesis but the implications of the findings are discussed in the concluding chapter.

Fire can be started from at least four sources (Glaves & Haycock, 2005), these are:

a. Prescribed burning, sometimes called managed or controlled burns, where the fire has been deliberately lit for management purposes. In the UK, prescribed burning is controlled by legislation (Table 1.3) and is specifically confined to between October and mid-April. Small patches are burned carefully on a rotational cycle, which can be as little as 7 years where there is very vigorous growth of *Calluna* up to as long as 25 years where growth is slow (Jenkins *et al.*, 1970). Wildfire – according to the definitions of Canadian Interagency Forest Fire Centre CIFFC, (2002) and the NWCG National Wildfire Coordinating Group (2008) are any unwanted or unplanned fires; these can be sub-divided into at least three types:
b. Escaped prescribed fire, where the fire has moved beyond the planned fire boundary and is out of control.
c. Human-induced fires started accidentally by negligence (barbecues, smoking, discarded glass) or started deliberately (arson).
d. Natural-fires. These are usually started by lightning ignition (Sugihara *et al.*, 2006), with a very small percentage started by spontaneous combustion of dry fuel such as sawdust and leaves (Scott, 2000). Although they can also be started after volcanic eruption and near wetlands through either respiration combustion and methane ignition (Beckage *et al.*, 2005).

(b) Response of species to fire

The responses of species to fire can be classified into four basic types (Hobbs *et al.*, 1984):

1. Species, which have a large store of, buried seed in the soil bank which can germinate when the above-ground vegetation is removed by fire. These species often are stimulated either by high temperatures (Bannister, 1965, 1965) or smoke (Tieu *et al.*, 2001). Such species often produce very large numbers of emergent seedlings immediately after burning.
2. Species which can recover by vegetative reproduction, either by resprouting from stem bases or from underground roots or rhizomes (e.g. *Pteridium aquilinum*).
3. Serotinous species (e.g. some *Pinus* and *Eucalyptus* spp.) which maintain seeds in their unopened cones until fire opens them. Such species often have flammable bark due to

high concentrations of resins and oils, and therefore, fire spreads quickly through the ecosystem (Bond & Midgely, 1995).

4. Species which have wide-scale dispersal and their seeds colonize burn patches quickly by wind dispersal (e.g. *Betula* spp. Hobbs *et al.*, 1984).

Table 1.3. Legislation covering the legal prescribed burning season in the UK (Clay *et al.*, 2010).

	Uplands	Lowlands	Principal legislation	Code
England	1st October – 15th April (SDA)	1st November – 31st March	The Heather and Grass etc. Burning (England) Regulations 2007	The Heather and Grass Burning Code (Defra, 2007a)
Wales	1st October – 31st March (SDA)	1st November – 15th March	The Heather and Grass etc. Burning (Wales) Regulations 2008	The Heather and Grass Burning Code for Wales (Welsh Assembly Government, 2008)
Scotland	1st October – 30th April (above 450m)	1st October – 15th April (below 450m)	Hill Farming Act 1946	Muirburn Code (SEERAD, 2001a) Muirburn code supplement (SEERAD, 2001b)
Northern Ireland	1st September – 14th April		Game Preservation Act (N.I.) 1928, Chapter 25 as amended by the Game Law Amendment Act 1951, Chapter 4	

1.5.2. Prescribed burning of moorlands in the UK

(a) The extent or prescribed burning in the UK

Data on the amount of prescribed burning that has occurred vary, and depend to a large extent on the measurement methods used, the timescale over which the assessments were made and the geographic area studied. A recent assessment of the amount of prescribed burning, for example, suggested that up to 114 km^2 of the UK uplands are burnt annually (Yallop *et al.*, 2006b). However, estimates of the amount of peat subject to some prescribed burning suggest that 1000 km^2 out of 6780 km^2 of deep peat, (Natural England (2010a), or 15% of English peatlands have been subjected to some prescribed burning, a similar percentage to the estimate for the UK of 46.5%, 3150 km^2, (Defra, 2010). However, the proportion of burning varied significantly across the UK: 1-2% in Borders and Grampian (Hester & Sydes, 1992) to 20% in the North Pennines AONB (Yallop *et al.*, 2006a). Yallop *et al.* (2005), using historical and current aerial photography, showed that 38% of the area was managed by burning in *Calluna*-dominated communities, but in areas are not normally managed for grouse, these proportions are much lower 1-16% (Glaves & Haycock, 2005).

Yallop *et al.* (2005) also showed using a national sample survey of aerial photography, that burning appeared to have increased between the 1940s and 1970s but there was little change between 1970 and 2000. In a subsequent study, using photography of the National Parks, a significant increase in burning was recorded from 1970 to 2000, demonstrating the localised regional variations in burning (Yallop *et al.*, 2006b). Indeed in a study of burning during the 1990s on the High Peak estate, Derbyshire, there were increases in both number of burns and total area burnt (Penny Anderson Associates, 2006). The authors suggest that these increases in Derbyshire were due in part to the Environmentally Sensitive Area (ESA) Ministry for Agriculture, Fisheries and Food (MAFF, 1993) agreements that grant-aided an agreed burning programme. However, increases and decreases are also influenced by the annual burning conditions and these authors noted a reduction between 1999 and 2005, possibly because of particularly wet years.

Recently the use of burning has been the subject of review (Tucker, 2003) and the consequence has been substantive debate on whether prescribed burning should be continued. The reason for this debate is that moorlands must not just contribute to an increasing range of conservation objectives, ranging from the conservation of species and communities but also provide a range of ecosystem services (carbon sequestration, water provision and recreation) (MEA, 2005; Marrs *et al.*, 2007; Harris *et al.*, in press). It is almost inevitable that burning will impact on water quality given that there will be increased run-off and nutrient outputs (Bormann & Likens, 1978). Information on the impact of burning is conflicting with some authors suggesting no significant differences on dissolved organic carbon (DOC) in soil and stream waters between burned and unburned sites (Ward *et al.*, 2007; Worrall *et al.* 2007a; Clay *et al.*, 2009b), and others (Yallop & Clutterbuck, 2009) maintaining that increased DOC concentrations were brought about by an increasing use of burn management. It is likely that losses of DOC will be proportional to fire intensity and they will be lower under prescribed burning management regimes and much greater under severe wildfire. Water quality is a major issue as there is a need under the EU Water Framework Directive (WFD) that inland waters achieve 'good ecological status' (i.e. good chemical, hydro-morphological and biological status) by 2015.

(b) *Techniques for prescribed burning of moorlands*

Prescribed burning in a mosaic pattern will maximise the productivity of the moor for sheep and grouse and at the same time reduce the overall biomass on the moorland and hence reduce the fuel load (Clay *et al.*, 2010). The patch size in the burn mosaic are often much larger when the moor is managed for sheep and deer than where grouse is management is the main focus (Gimingham, 1972). If a moor managed in this way is subject to wildfire it is predicted that the fire will be more easily controlled by the Fire Service, and will have a lower intensity, hence that there will be less loss of vegetation and soil organic matter. Moreover the burned vegetation will be more likely to recover quickly via resprouting from the burned stems and germinating seeds from the surface soil. Unfortunately, the optimal prescribed burning frequency to minimise wildfire risk is not known.

The intensity of the fire produced during prescribed burning will depend on a range of variables, e.g. the time of year, prevailing weather conditions, and the age of the stand at time of burning. Autumn burns usually produce lower intensity fires than spring burns, since the vegetation and litter have higher moisture content in the autumn (Hobbs & Gimingham, 1987). Similarly, intensity of burning may be lower if preceded by wet weather. The intensity of the fire will also affect vegetation regeneration. This may interact with other variables, for example Miller & Miles (1970) argued that *Calluna* regenerated more successfully after autumn fires, partly because potentially competitive species such as *Eriophorum vaginatum* and *Trichophorum cespitosum* were less vigorous at this time of year (Mowforth & Sydes, 1989).

Where the aim is to maximize *Calluna* regeneration, prescribed burning should ideally remove all above-ground vegetation, including moss (Davies *et al.*, 2010), whilst minimizing damage to roots and soil. This is equivalent to "light" fire severity, where surface litter, mosses and herbs are charred or consumed but the soil organic layer remains largely intact. Prescribed burns in the UK are traditionally lit using a naked flame from a wick or "fire kettle". However, this approach only works well with relatively dry vegetation and is thus limited by weather. The number of days that burning can be carried out is, therefore, limited, and even when burning is possible, burns can only be started late in the day. More recently, an approach has been developed where pressurized diesel or sprayed on to the ignition site before lighting. This technique has been referred to as "cool" burning" but it is perhaps better described as "pressurized-fuel-assisted", or "PFA" burning. This technique was developed to achieve biomass removal without damage to underlying peat and leaving the bryophyte layer charred, but otherwise intact. The PFA burning technique allows prescribed fires to be lit during wetter conditions, and hence increases the number of days within the burning season when burning can be implemented. Regeneration occurs from resprouting *Calluna* within the remaining bryophyte layer within the first summer. Where the bryophyte layer is wholly removed, regeneration tends to be slower (G. Eyre, unpublished). Nevertheless, the aim is to achieve removal of the above-ground vegetation; the *Calluna* should then regenerate quickly by resprouting from the burnt *Calluna* stems and also from seed.

With wildfire the impact of the fire is much more variable, ranging from results similar to a prescribed burn in terms of fire intensity (light, removal of above-ground biomass) through to exceptionally severe (removal of biomass and all underlying organic material). Prescribed burns in the UK must be done outside the driest part of the year and this aids fire control. Wildfire, on the other hand, can occur at any time of the year and if this occurs in very dry conditions produces very hot fires which are uncontrollable. In the drought year of 1976 very hot fires caused considerable damage on the North York Moors (Maltby *et al.*, 1992) which burned for a very long time (months) as the peat kept re-igniting and was completely removed in some places, leaving mineral soil. Where this occurred, regeneration was extremely slow (Legg *et al.*, 1992).

(c) Rotational extent of prescribed burning in current moorland practice

The frequency at which the vegetation should be burned depends very much on local conditions and the result required. For wildlife conservation and to promote biodiversity, a burning rotation of 12-15 years has been suggested (Hobbs & Gimingham, 1984) as it had a less detrimental impact on the developed flora and fauna than shorter rotations. On the other hand, a shorter rotation might mean that *Calluna* will still be in the building phase when burnt (Paterson & Miller, 1991), providing a continuous abundance of the young and nutritious shoots which are a priority for grouse management. However, Watson & Miller (1976) recommend that the optimal interval between burning should be determined by the response of the heather, rather than a specified interval, ideally when it reaches 20-30 cm in height, *i.e.* in the late building phase (MAFF, 1992). Much of this variation is dependent on the local growth rate of the vegetation and both the Muirburn Code and Burning Code highlight the need for burning rotations appropriate to the local conditions (Defra, 2007a; SEERAD, 2001a) with shorter rotations required in the warmer southern parts of the UK, and perhaps longer rotations further north

1.6. The Peak District as a regional moorland area within the UK

The Victorian art critic and pioneer conservationist John Ruskin described Derbyshire In "Arrows of the Chase" as "*a lovely child's alphabet; an alluring first lesson in all that is admirable* in the way it engages and fixes the attention" (Ruskin, 1840 to 1880 reprinted, Ruskin, 2000).

The Peak District straddling Derbyshire and Yorkshire is a largely upland and still a mostly uncultivated landscape. The current landscape reflects the underlying geology, which divides the area into the "White Peak" on limestone and the "Dark Peak" on millstone grit and shales. The lack of intensive cultivation means that there is much historic evidence of human settlement from the Mesolithic era through to modern times (Smith, 2009). The "Dark Peak" (which includes most of the moorlands) has a relatively cool and wet climate. The nearest meteorological station, Harper Hill, near Buxton, National Grid Reference SK 057708, altitude of 370 m, had a mean temperature of 10.3 °C and a rainfall of 1,025 mm in 1999. The temperature is similar to the average for England and Wales but the rainfall is above average (Peak District National Park, PDNP, 2000). However, most "Dark Peak" moorland is 300 m or more above sea level and the rainfall is much greater. Unfortunately, we have few data on weather conditions on the moorlands themselves. However, winter can be harsh and in the 1970s, there were regularly over 70 days when snow was recorded. In recent years there have been fewer days with snow lie, possibly as a result of climate change caused by global warming. Despite this there are still frosts on the moors for 25-30% between October and February.

The vegetation history reflects that described above (Section 1.2). After the last ice age (ca.15, 000 BP), pollen diagrams probably a forest of Scots pine (*Pinus Sylvester's*) and birch (probably mostly hairy birch, *Betula pubescent*) with an understory of *Calluna* and *V. myrtillus* in dry areas with *Salix* spp. scrub in wetter areas (Conway, 1947, 1948, 1949; Tallis, 1964a). Logs have been found at the mineral soil/peat transition on Bleak low, but there is little direct evidence of trees at greater elevations (320 m) on the South Pennines, including Featherbed Moss, which is part of Howden moor, one of the study sites used here (Tallis, 1964a).

The formation of sub-fossil wood indicates that conditions at the time were wet and acidic and ideal for preservation in peat (Tallis, 1964b). Progressive leaching of the mineral poor soils presumably led to the build up of acid humus and, with the onset of a wetter climatic period around 7,000 BP, led to the growth of *Sphagnum,* presumably initially in hollows, and the beginnings of the build up of a highly mummified peat with much *Eriophorum vaginatum* (tussock cotton grass) with heather and crowberry. The timing of the beginning of peat formation corroborates Conway's (1948) conclusion that peat formation began in "Atlantic times".
The vegetation history has been chronicled by Tallis (1964b, c) as follows:

(a) ca. 7,000-3,200 BP, the onset of a cooler climate coincides with the beginnings of the development of blanket bog due to the growth of *Sphagnum.* This is correlated with a decline in *Calluna* and an increase in sedges (presumably largely *Eriophorum* spp.).

(b) ca 2,600 BP, the peat development slowed and there was with an increase in *Calluna*, *Emporium* spp. and *Betula* spp., but the *Betula* pollen present may be from lower altitudes.

(c) ca 1,900 BP, there was a marked decline in *Sphagnum* and *Eriophorum* spp. in the peat.

(d) ca 1,600 BP, regeneration of the *Sphagnum* bog growth correlates with climatic conditions that favour peat develop (cool, wet climate).

(e) ca 1,000 BP, there is another transition to a drier, more mummified peat with increasing *Eriophorum* spp. and the presence of less tree pollen and some *Plant ago* spp. pollen-suggesting disturbance or cultivation in the general area. Viking settlements have been found in Renton in Derbyshire which were dated to ca. 870AD.

At the present time the surface peat is highly mummified and there is very little current peat formation, so precise dating of the upper layers is difficult. However, soot deposition and the further decline of *Sphagnum* is particularly noticeable and presumably occurred mainly after about 1800 AD. Keepers and shepherds report significant *Sphagnum* presence of in the early 1900s, while Tallis (1964c) reports its almost total absence by the 1960s.

The Peak District is fortunate in that one of the pioneers of British plant ecology mapped and described the vegetation under the aegis of the British Vegetation Committee at the end of the 19^{th} and beginning of the 20^{th} century (Moss, 1904, 1913). Moss (1904) described the Peak District moors as "*a valuable asset which is turned too little account. Grouse are driven and shot over them, it is true; but considering the enormous rents paid by tenants for good grouse moors, it is surprising that more attention is not paid to the better cultivation of the heather and bilberry, as these plants are much better adapted*". The evidence presented by Moss provides good broad-scale evidence of change and in particular provides a baseline to show a large decline in bryophyte species and especially *Sphagnum* spp. in the twentieth century (Tallis, 1964b). Unfortunately the data presented by Moss are at the regional scale, and whilst he produced quite detailed maps, they are provided at the plant association stage, and it is impossible to derive detailed information for assessing change at the moorland scale, the scale studied in this thesis, except in very broad terms.

Since the industrial revolution there have been substantial changes within the area. The abundant water power was used as energy by pioneering industrialists who developed water-powered textile mills; this important development was recognized by the designation of the Derwent Valley Mills as a World Heritage Site in 2001. At the same time the Peak District is surrounded by major industrial towns and cities that grew to prominence during the industrial revolution (e.g. Manchester, Salford, Stockport, Sheffield, Derby, Rotherham, Chesterfield) and as such was subject to atmospheric pollution (Tallis, 1995), and high lead concentrations are now detected in the surface peats, which could cause significant problems of contamination of drinking water should surface erosion occur (Rothwell *et al.*, 2005; Merton, 1970; Davies, 1974; Lee & Tallis, 1973).

The dawn of the railway age during Victorian times heralded the start of the tourism industry in the Peak District, with a spa town at Buxton. Ruskin in his letters 'Arrows of the Chase' (Ruskin 1860, reprinted Ruskin, 2000) was less enamoured *"now every fool in Buxton can be at Bakewell in half an hour, and every fool in Bakewell at Buxton, which you think a lucrative process of exchange – you fools everywhere"*. With increase in the populations of the large surrounding cities, the working classes gained more leisure time, and there was an increased pressure to gain access to the uplands of the Peak District. The landowners resisted this pressure for access and in 1932 there was an organized mass trespass on Kinder Scout after which five ramblers were imprisoned. This pressure not only increased the demand to allow access to the moors, but also led to the designation 19 years later of the Peak District as National Park (Fig. 1.3). The Peak District because the United Kingdom's first National Park in 1951 following the implementation of the National Parks and Access to the Countryside Act (1949).

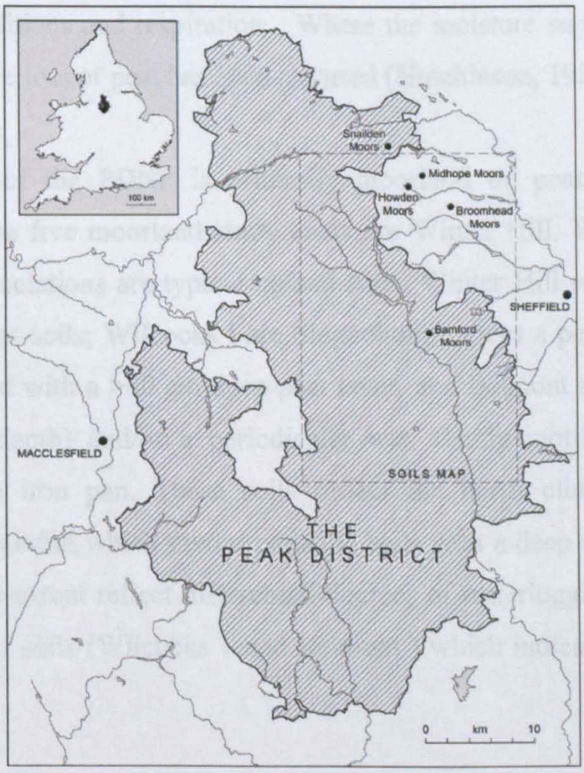

Fig. 1.3. Area designated within the Peak District National Park (shaded) and the five main moorlands study sites, which are in the northerly "Dark Peak area of the Peak District (drawn S. Mather).

National Parks provide some landscape protection in Britain. However, unlike National Parks elsewhere they are landscapes where many people live, work and visit. They receive some funding from central Government, but the National Park Authorities work with other agencies to attempt to safeguard the National Parks for the future. In total, National Park Authorities received £42 million in 2005 from the Department for Environment, Food & Rural Affairs. The Peak District National Park Authority (PDNPA) received £7.5m of that fund in 2005. The PDNP is still surrounded by major areas of population and 54.3% for the UK population live within a 50 mile radius of the park. There were an estimated 21.4 million visitors in 2009 (PDNP, 2009).

The PDNP (PDNP, 2009) have identified the major threats to the ecosystems in the park. The first is climate change associated with global warming, with both direct and indirect effects. The direct effects will be through a potential changing species complement. The indirect effects identified included an increase in summer droughts and hence a greater risk of moorland fires which will damage blanket bog vegetation and lead to peat loss, and the possibility that some birds (including moorland birds like golden plover) will breed earlier, when their insect food supply may not be available. They also suggest that the Peak District National Park may be unable to support blanket bog by 2050 because of combinations of increased temperature and extended periodic drought conditions predicted as a result of global warming as a result of an increase in aerobic conditions and respiration... Where the moisture supply of lowland fens has been reduced, substantive loss of peat has been reported (Hutchinson, 1981).

The "Dark Peak" are of the PDNP is primarily moorland on peat; the main upland soil associations found in the five moorland study areas are Winter Hill, Wilcocks 1 and Belmont (Fig. 1.4). All three associations are typical upland soils: Winter Hill soils are thick, very acid, perennially-wet, raw peat soils; Wilcocks 1 are Stagnohumic gleys a peaty topsoil; intermediate between stagnogleys and with a >40 cm deep peat layer, and Belmont are stagnopodzols with a peaty topsoil (15 cm depth) and/or a periodically wet, faintly mottled bleached subsurface horizon, often with an iron pan. These soils reflect the harsh climatic conditions of low temperatures and high rainfall which favour podzolic soils with a deep mor layer and peats. The soil differences to some extent reflect differential degrees of waterlogging from perennially wet (Winter Hill) to the gley soils (Wilcocks 1 and Belmont) which indicate differential degrees of waterlogging.

The moorland is owned by a variety of individuals and organizations. The management objectives for these estates vary among management units, and most have some form of designation or support mechanism (ESA prescriptions) to help conserve the moorland vegetation. Prescribed burning is part of the management applied on most moors, and in the PDNA with its high visitor pressure, all have a high summer wildfire risk. For example, on Bleaklow in April 2003, there was a severe wildfire (MFF, 2006, Fig. 1.5a) which affected an area of ca. 850 ha of land. In the burned area there has been substantial peat loss and where vegetation regenerated naturally after the fire this has been slow (Fig. 1.5b).

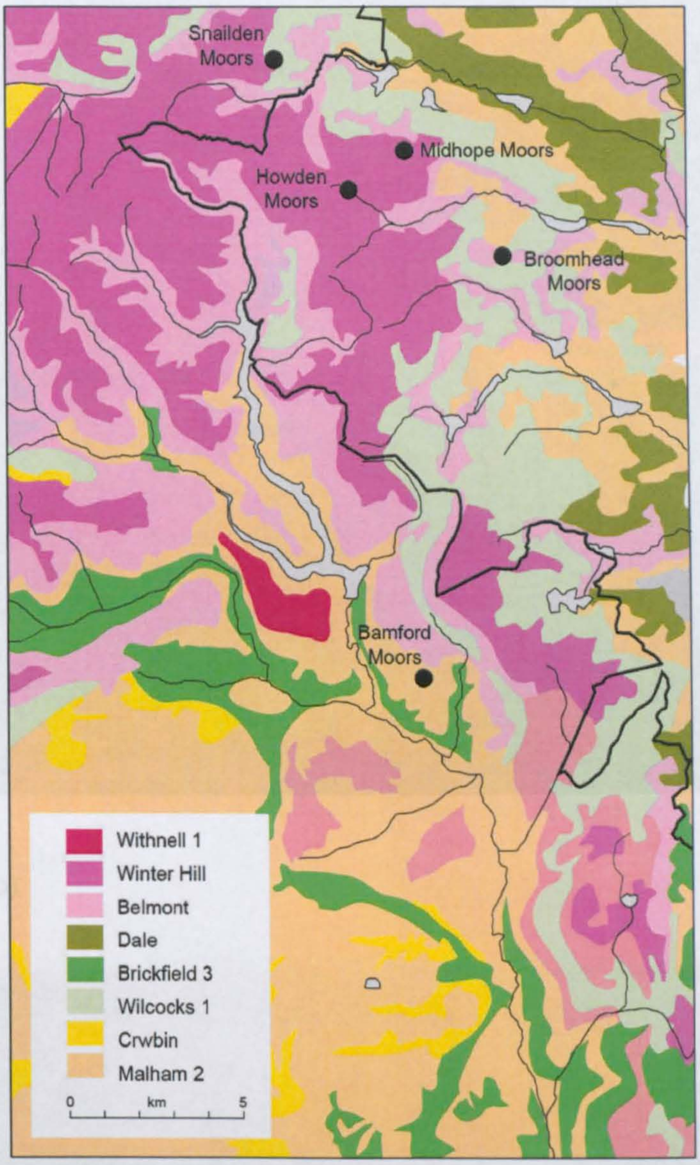

Fig. 1.4. Soil series map of the Dark Peak area of the Peak District showing the five moorland study sites (Map produced by S. Mather using data from the National soil series map produced by the National Soils Resources Institute, Cranfield University).

(a)

(b)

Fig. 1.5. (a) Photograph from space showing the smoke plumes from wildfires in 2003. Bleaklow is marked with an arrow (photograph courtesy of Defra), (b) slow vegetation recovery at the latter site with extensive bare peat after six years (Aylen, 2009).

1.7. Aims of this study

The overall aim of this thesis is to develop a better understanding of the role of prescribed burning in moorland management in the Peak District. At present there is considerable conflict regarding the role of prescribed burning in the Peak District. On the one hand there is a need to maintain and improve the existing moorland vegetation to meet Biodiversity Action Plant targets Anon (1995a, b), whilst on the other there is a plan to use the peatland carbon store to help meet carbon sequestration and emission targets (Worrall *et al.*, 2009; Clay *et al.*, 2010) and water quality objectives (Yallop *et al.*, 2006; Yallop & Clutterbuck, 2009). Prescribed burning will clearly impact on all of these, and here is very little information to guide future policy that is evidence-based. Accordingly, this thesis intends to answer the following questions:

(a) How degraded are the moorlands in the Peak District, and does prescribed burning affect species density and restoration potential? (Chapter 2).
(b) What are the environmental factors that influence the response of the plant communities, and how do the constituent species respond after prescribed fire? (Chapter 3).
(c) Does prescribed burning affect soil chemical properties? (Chapter 4)
(d) What factors affect biomass reduction in prescribed fires on upland moorland? (Chapter 5).
(e) What changes in above-ground biomass, carbon and nitrogen occur during prescribed burning? (Chapter 6).

Finally, the main implications from this study are then brought together in a final discussion on the future of British moors and the role of prescribed burning (Chapter 7).

Chapter 2

Restoring moorland vegetation on peat: its potential and the role of prescribed burning

2. Restoring moorland vegetation on peat: its potential and the role of prescribed burning

2.1. Introduction

The conservation of nature is becoming increasingly complex. On the one hand legislation has been implemented to comply with the UN Convention of Biological Diversity at both European Union (Anon, 2006) and national level (Anon, 2007) with ambitious targets of halting the decline of biodiversity in the EU by 2010 and to restore habitats and natural systems (Anon, 2006). At the same time the importance of natural systems for the provision of ecological services was highlighted in the Millennium Ecosystem Assessment (Anon, 2005). The implication of these policies is that habitats that have been degraded are at worst to be maintained at their present level of biodiversity, and if possible, to be improved in terms of their biodiversity status, which will hopefully provide better ecosystem services. Set across this policy framework is an increasing realisation that conservation action (essentially expenditure) should be evidence-based (Pullin & Knight, 2009).

Clearly if the general aim of halting biodiversity decline and restoring natural systems is to be implemented, then the general philosophy of ecological restoration should be used (Bradshaw, 1983; SERI, 2006). This requires the designation of a "target community" to which the restoration can aspire. Where land has been severely damaged and remnant vegetation of high conservation value still exists nearby then this is relatively easy to implement (Aronson *et al.*, 1995; Holl, 2002). Indeed, where damage has been more substancial through successional change, and management is required to modify vegetation, techniques have been developed to identify the scale of the restoration required and monitor its progress (Mitchell *et al.*, 2000; Fagan *et al.*, 2008). However, where large landscapes have been degraded over time through a variety of damaging factors, it is harder to develop a strategy, because the entire landscape has been affected there are no good reference sites against which to judge success. Moreover, where the landscape is a cultural one, which has been managed by humans for millennia, any restoration needs to be sensitive to the requirements of current land managers. Where such a landscape is also highly valued for the provision of important ecosystem services, the complications increase (Marrs *et al.*, 2007).

In this chapter, we consider the restoration potential of upland moorland in the Peak District National Park in central England where all of these issues occur. Moorlands in the UK have been designated as of high international conservation significance, with 75% of the world's moorlands occurring in the UK (Thompson *et al.*, 1995; Ratcliffe

& Thompson, 1998) and with six vegetation communities that are almost restricted to the UK (Thompson *et al.*, 1995). Moorlands are dominated by dwarf shrubs, mainly *Calluna vulgaris* (L.) Hull (Gimingham, 1972), and many are managed for sheep grazing and game management (red grouse, *Lagopus lagopus scoticus* (L.). Usually this management includes prescribed burning on rotation, which produces a patchwork mosaic of vegetation in different stages of the burn-recovery cycle. Approximately, 65% of British upland moors are managed using this mosaic approach for the benefit of red grouse (Sotherton *et al.*, 2009). Prescribed burning removes woody plant growth and any old degenerate plants, leaving charred stems. The burnt stems resprout to produce new growth, although regeneration reduces with plant age, falling off sharply after ca. 15 years (Lovat, 1911; Miller & Miles, 1970). Where resprouting is slow, plants must regenerate from seed (Gimingham, 1994). Prescribed burning has been used for at least 200 years and possibly much longer (Pearsall, 1980; Birks, 1988; Simmons, 2003). The moorlands in the Peak District National Park have been affected very severely by aerial pollution (S and N compounds) from surrounding industrial cities since the mid 1700s (Tallis, 1964, 1998; Ferguson & Lee, 1983), overgrazing (Anderson & Yalden, 1981), and intermittent, but very damaging wildfires (Albertson *et al.*, 2009; Maltby *et al.*, 1990). However, as the impact of atmospheric pollutants is declining, it is hoped that species diversity in these moorlands should start to recover.

In the last few years there has been some debate over moorland management techniques (Tucker, 2003). The differing opinions vary between users and stem from different overall objectives for the desired management, for example: (a) for conservation a bog vegetation dominated by graminoids and bryophytes may be preferred to a *Calluna*-dominated one; (b) users and potential users of ecosystem services have suggested that prescribed burning leads to a reduced carbon store and impaired water quality with greater clean-up costs (Worrall *et al.*, 2009; Yallop *et al.*, 2006; Yallop & Clutterbuck, 2009; Clay *et al.*, 2010), and (c) some have speculated that the long-term response of moorland vegetation to the cessation of prescribed burning will be either a graminoid/bryophyte-dominated vegetation or a more species-rich one. There are, however, very few data on (1) the state of the moorland vegetation in the area, (2) a target reference community in the region against which to measure success, because the damaging pressures on the moorland acted at the regional scale and most of the moorland is viewed as having unfavourable status, and (3) the response of the current vegetation to prescribed fire. Indeed, in a systematic review was assessed the impact of burning as a conservation intervention on heaths and bogs, the outcome was that evidence was insufficient to generate robust management recommendations (Stewart *et al.*, 2005). Moreover, if the aim is to enhance the species diversity of moorland vegetation with unfavourable status, there will almost certainly be a need for new species to colonize. The obvious source for these new species is the propagule bank present within the ecosystem (Thompson, 1992; Bekker *et al.*, 1997; Pakeman & Hay, 1996; Pakeman *et al.*, 1998; Miller & Cummins, 2003; Ghorbani *et al.*, 2003). It is well known that *Calluna* has a large buried, long-lived seed bank (Chippendale & Milton, 1934), but there is little information on many of the other species in these moorlands. The seed bank is dynamically linked to the species present, although some may be present in one, and not in the other (Thompson & Grime, 1979).

Accordingly, in this chapter we set out to assess the state of both the vegetation and propagule bank in a selection of moors in the Peak District National Park. This study used a space-for-time substitution or chronosequence approach with respect to time since prescribed burning. In a recent review on fire management for biodiversity conservation this approach was highlighted as having considerable merit especially if pseudoreplication was accounted for (Driswcoll *et al.*, 2010). In this study, therefore, sampling was replicated across different moors to provide some degree of assessment of spatial effects and variability in management practice (n=5 for vegetation; n=3 for propagule banks), with stratified random sampling of patches of different ages within each moor. Here, good information was also available on past management from estate management archives. This survey provided the first assessment of moorland species composition at the regional scale, and hence provided, in the absence of an historical datum, a current baseline for assessment of recovery from past damage. As there were no historic baseline or reference sites available which could be used as a guide to assess degree of damage/recovery statistically, the species composition and richness of these moorlands were assessed in relation to species composition in idealised target reference communities National Vegetation Classification (NVC), (Rodwell, 1991, 1992). The NVC communities are communities of semi-natural vegetation found in areas of high conservation value in the UK, and the variation detected provided a crude measure of β-diversity. Three hypotheses were then tested with respect to the species composition of the vegetation:

Hypothesis 1: Prescribed burning has no effect on plant species richness. If burning affects species richness then there are two alternate hypotheses.

Hypothesis 2: Prescribed burning has a positive effect on plant species richness.

Hypothesis 3: Prescribed burning has a negative effect on plant species richness.

Thereafter, the recovery potential of species from the propagule bank was assessed by describing the species composition (higher plant seeds, bryophyte spores and fragments in both litter and soil seed banks). The effects of prescribed burning were tested on the most common species.

2.2. Methods

2.2.1. Study areas and sampling protocol

The species composition of the vegetation was studied at five moorland management units (Bamford, Broomhead, Howden, Midhope, and Snailsden Moors) in the North Peak ESA, within the Peak District National Park, UK over three year beginning in 2006 (Table 1). All of the burned patches on each moor were mapped using aerial photography taken in September 2005 and cross-referenced with land burning management maps provided by the land managers. This provided a series of burned patches of known age (elapsed time since burning) between 2-16 years; older patches were also identified where no burning had been carried out for at least 35 years. There were 79, 249,103, 271, and 252 available burn patches on the five moors respectively. For vegetation surveying, ten burns from each moor were selected using an age-stratified random sampling procedure over the three survey years; only two moors were sampled in 2007, and the older patches were sampled only in 2007 and 2008 (Table 1). The propagule study was carried out in 2009 on Bamford, Broomhead and Howden Moors. Here the same sampling strategy was used to randomly select ten patches that had been burned at varying times in the last 20 years, plus three older patches.

Once selected, the geo-referenced outlines of each of the sampled patches were digitized from aerial photographs, and their areas calculated within (ArcGIS, 2009). The number of potential $1m^2$ quadrats available within each burn patch was counted and a random selection made for field sampling (Quadrats with patches; n=10 in 2006 and 2008, n=4 in 2007, total n=1010; propagule study n=4 in 2009, total n=156).

2.2.2. Vegetation survey

On each of the five moors, the patches and then the quadrat positions were located using GPS (eTrex Venture® HC). The cover of all higher plants, mosses and lichens were recorded and vegetation height measured by placing a disc (diameter area=0.30 m, mass=0.2 kg) in the centre of every quadrat and measured as distance (cm) from the disc to the ground Stewart *et al.* (2001). Formal quality control procedures were used at all stages of quadrat selection, and

Table 2.1.
Details of the five study moorlands in the Peak District National Park, UK where the post-fire vegetation succession was surveyed.

Moorland site	Longitude & Latitude	British National Grid squares digitized per Moor	Elevation range (m)	Age range (yrs)	Estimated age of older patches (yr)	Sampling years
Bamford	Latitude 53°21'N, Longitude1°40'W	SK 1993, 1994, 2093, 2094	300-420	3-14	38	2006, 2007, 2008
Broomhead	Latitude 53°27'N, Longitude1°38'W	SK 2395, 2394, 2295 2294	300-460	2-15	40	2007, 2008
Howden	Latitude 53°28'N, Longitude1°42'W	SK 2184, 2185, 2284, 2285	272-540	2-15	50	2006, 2007, 2008
Midhope	Latitude 53°29'N, Longitude1°40'W	SK2198, 2197, 2099, 2098, 2097, 1999, 1998, 1997	270-480	3-15	40	2007, 2008
Snailsden	Latitude 53°30'N, Longitude1°44'W	SE1503, 1501, 1404, 1401, 1400, 1304	350-470	3-16	50	2007, 2008

in all aspects of the field work. Nomenclature follows (Stace, 1997) for higher plants, (Atherton *et al.*, 2010) for bryophytes, and (Dobson, 2000) for lichens. A soil sample was taken and the depth of peat measured if depth was < 50 cm. On four moors all quadrats had a peat depth of at least 50 cm, the exception being Midhope where some quadrats were on shallower peat (20-30cm =4.3%; 30-40cm=14.2%, 40-50cm=3.7%; >50cm = 77.8%).

2.2.3. Assessment of propagule distribution

Four 1 m^2 sampling random positions were located in each patch and the vegetation from the central 25 x 25 cm removed, cut and separated into living material and litter. Five soil cores (5cm diameter, 7cm depth) were taken from each corner of the 1m^2 quadrat and the centre, pooled, weighed and stored at ~5°C before processing.

Seed composition and density were estimated in the entire litter fraction and a soil sub-sample of known mass from each quadrat. The sampled material was decanted into separate plant growing trays (20.5 cm x 15.5 cm) lined with a mesh net (1mm) and filled with layer of washed, horticultural sand. Control trays were included to identify seeds of foreign species which may have been present in the glasshouse. Two types of control trays were used, providing different physico-chemical properties for germination, containing either sand alone, or peat collected from Howden Moor that had been sterilized by autoclaving. The few individuals that were detected in these control trays were non-moorland species and were discounted in the analyses. The trays were set out on two benches in an unheated glasshouse. Both benches were covered with plastic sheets and watered from below using capillary matting (2 x 30 minute watering cycles per day). This maintained a good water supply from below to the developing seedlings, but they were also watered manually from above at least twice weekly. All seedlings establishing from both litter and soil samples were identified and counted after 10 and 20 weeks. Where identification was uncertain at the time of counting, the seedling was transplanted into a separate pot and grown on, so that an accurate identification could be made. The seedling densities per quadrat were corrected to numbers m^{-2}. Since it is difficult to measure emergence of individual mosses from soil banks, because they can be derived from spores or extant leaf fragments, mosses were recorded on a presence/absence basis only for each sample.

2.2.4. Data analysis

(a) Assessment of the current state of the vegetation

In order to assess community β-diversity and quality of vegetation on each moor, every quadrat (n=1010) was allocated a British National Vegetation Classification (NVC) community type (Rodwell, 1991, 1992) using TABLEFIT (Hill, 1996). The NVC class with the greatest composite Goodness-of-fit score was selected for each quadrat G-score, (Hill, 1996). The most common NVC classes detected were then chosen as the target reference community types, and the goodness-of-fit provided an assessment of how close the moorland vegetation was to these targets. In addition, three crude, qualitative assessments were made of recovery status. To do this, the species lists and frequencies of the most common NVC classes were inspected (Rodwell, 1991, 1992) and those species that do not occur in the region were removed checked against (Dobson, 2002; Preston *et al.*, 2002; Atherton *et al.*, 2010). The species were then grouped into three: (1) Absent-members, species in the NVC list but absent in this survey; (2) Extras, species detected in this survey but not in the NVC list (essentially weeds), (3) Present-members, species that were present in this moorland survey and the NVC classes. The Absent-members were sub-divided into those species that were noted in a walk-over survey of the Moors, and those that were not.

(b) Assessing the response of plant species diversity to prescribed burning

All analyses were implemented in R software v. R-2.11.0, (R Development Core Team, 2010), using the vegan package (function 'diversity' for calculation of diversity indices (Oksanen, 2010), and the lme4 package for Linear Mixed Models (function 'lmer') (Pinheiro & Bates, 2002; Venables & Ripley, 2002).

Four measures of species diversity were calculated for each of the 1010 quadrats: these were species richness (total number of species), and the Shannon-Weiner, Simpson's and Evenness indices (Krebs , 1972). Each of these measures of diversity was then analysed using mixed-effects models (Crawley, 2007) with a Poisson error distribution. Both moor and elapsed time since burning were included as fixed factors in the model, and the diversity of patches nested within each moor were treated as random effects. This approach accommodated the hierarchical nature of the study design, and allowed five models to be tested using the stepwise deletion approach to derive the Minimum Adequate Model (MAM), (Crawley, 2007); the models were (i) the interaction model (moor x elapsed time), (2) the additive model (moor + elapsed time), the moor only model, the elapsed time only model, and the null model. Terms were deleted sequentially using the AIC statistic (Crawley, 2007).

Although great care was taken to ascertain the age of the very old patches, there is greater uncertainty of their exact age, and this might clearly influence the shape of any response curve. To ensure that any detected response was valid, the analyses were repeated using vegetation height in place of elapsed time since burning. Vegetation height was measured at each quadrat, and hence is not subject to the same uncertainty as elapsed time since burning. Here, moor was included as a random variable.

Results from all four measures of diversity showed the same trends, so only the data for overall species richness are presented here. Sub-analyses were also carried out on both sub-groups of species (a) higher plants, and (b) mosses and lichens.

(c) Assessing the response of propagule distribution to prescribed burning

The effects of burning on the abundance of the most common species, (present in > 10% of samples, n = 16) were analysed using mixed-effects models as described above. A Poisson error distribution was used for the densities of higher plants and a binomial one for the bryophyte presence/absence data.

2.3. Results

2.3.1. Assessment of the current state of the vegetation

The community β-diversity was assessed by counting the number of NVC communities/sub-communities that were found on each moor, Twenty-eight NVC community classes/sub-classes were found across all moors; Bamford had all 28, Howden was intermediate with 15 and the others all had 9 (Table 2.2). The goodness-of-fit values are typical of communities that are not pristine, they range from a poor fit (<50%) through to some communities where there is a considerable range of fit with some quadrats with a very good fit (e.g. H1e sub-community = maximum of 96% fit). Fifteen of the 28 classes detected had some quadrats with a goodness-of fit greater than 50%. When the sub-communities were subsumed into the overall community class the difference between moors disappeared with 17 classes at Bamford and Howden and 16 at the others. Nevertheless, most of the quadrats were classified into seven main NVC communities, four heathland classes (H1, H9, H11, H12), two mire classes (M19, M20) and an upland grassland community (U2).

Sixteen species typical of the most common NVC classes detected on the moorland were present, and most were present at frequencies more or less similar to the NVC typical values Table 2.3, (Rodwell, 1991). Six species were detected which were not in the NVC lists and three of these (*Juncus effusus*, *Campylopus introflexus* and *C. pyriformis*) might impede the

Table 2.2.
National Vegetation Classification communities detected on the five study moorlands in the Peak District National Park, UK between 2006 and 2008. Mean values and ranges of the Goodness-of-fit statistic provided by Tablefit (Hill, 1996) are presented. Where no range is given (n=1), communities denoted with an asterisk were found in less than 0.1% of sampled quadrats. Shaded data indicates at least some Goodness-of-fit values >50%.

NVC community/ sub community	No of quadrats	Goodness-of-Fit		NVC description	
		Mean	Min-Max	Main community	Sub-community variant
H1*	1	64	-	*Calluna vulgaris-Festuca ovina* heath	
H1c*	4	46	43-49		*Teucrium scorodonia*
H1e	242	72	37-96		Species-poor variant
H8e*	1	24	-	*Calluna vulgaris-Ulex gallii* heath	*Vaccinium myrtillus*
H9*	7	62	51-71	*Calluna vulgaris-Deschampsia flexuosa* heath	
H9b	34	58	31-67		*Cladonia* spp.
H9c	53	62	48-80		Species poor variant
H9d*	10	47	17-70		*Galium saxatile*
H10*	3	51	50-51	*Calluna vulgaris-Erica cinerea* heath	
H11*	2	56	56-56	*Calluna vulgaris-Carex arenaria* heath	
H11c	117	63	23-74		Species poor variant
H12	15	55	29-70	*Calluna vulgaris- Vaccinium myrtillus* heath	
H12a*	8	63	51-74		*Calluna vulgaris*
H12c*	7	57	48-66		*Vaccinium myrtillus –Cladonia impexa* heath
H18c*	1	52	-	*Vaccinium myrtillus-Deschampsia flexuosa*	
M2*	8	55	36-76	*Sphagnum cuspidatum /recurvum* bog pool	
M6c*	1	13	-	*Carex echinata-Sphagnum recurvum/auriculatum*	*Juncus effusus*
M16*	3	35	28-39	*Erica tetralix-Sphagnum compactum*	
M19	47	50	29-75	*Calluna vulgaris-Eriophorum vaginatum* blanket bog	
M19a	76	64	48-83		*Erica tetralix*
M19b*	4	45	32-61		*Empetrum nigrum*
M20	188	67	32-90	*Eriophorum vaginatum* blanket/raised bog	
M20a*	2	31	28-33		Species poor variant
M20b	138	69	37-87		*Calluna-Cladonia*
U1e*	2	37	30-43	*Festuca ovina-Agrostis capillaris-Rumex acetosella* grassland	*Galium saxatile-Potentila erecta*
U2	16	53	17-84	*Deschampsia flexuosa* grassland	
U2b	11	66	38-88		*Vaccinium myrtillus*
U5d*	9	44	28-55	*Nardus stricta-Galium saxatile* grassland	*Calluna vulgaris-Danthonia decumbens*

regeneration of other species. Moreover, there was a large list of species (n=47) that are present in the NVC lists and might be expected to present in the area that are missing from these moorlands. Seventeen of these missing species were detected in a walk-over survey, implying that there are species present that occur below the detection limits of this survey.

2.3.2. Assessing the response of plant species diversity to prescribed burning

For both elapsed time since burning and vegetation height the interaction model was the MAM (Table 2.4), and the reduction in AIC from the null model was 19% and 28% respectively. Species richness showed the same responses (Fig. 2.3). Bamford had the greatest species richness (highest intercept) followed by Howden, Snailsden, Midhope and Broomhead. On all moors there was a reduction in species richness with respect to both elapsed time since burning and vegetation height. The reduction was greatest at Bamford and Howden, intermediate at Midhope and Snailsden and least at Broomhead. However, the important results are that (a) there is a measured decline in species richness as the vegetation ages after burning or as the vegetation increases in height on all moors, and (b) there is no evidence of an increase in species richness. The greatest diversity was found on patches burned less than 15-10 years previously and with vegetation below 50 cm in height. The analyses of the higher plants and bryophytes+lichens yielded similar conclusions so they are not discussed in detail. The model fits, albeit significant, were not as good in terms of AIC reduction.

However, because these conclusions could be criticised due to the gap in the temporal record and the fact that the old sites may be atypical, the analyses were redone with the old sites removed. In these analyses, elapsed time was not significant and there was only a significant difference between moors (Table 2.4). However, the relationship with vegetation height was almost exactly the same as those with all data included, except that only three sites produced significant interactions (Bamford, Broomhead, and Midhope), Bamford and Midhope showed negative relationships and Broomhead showed a positive one with respect to vegetation height over the reduced range (Fig. 2.1b).

2.3.3. Assessing the potential role of the propagule bank for restoration its interaction with prescribed burning

Of the higher plants only *Calluna* and *Juncus effusus* were detected at a frequency large enough to analyze statistically (present in > 10% of samples, n>16, (Table 2.5). *Agrostis capillaris, Agrostis vinealis, Carex binervis, Empetrum nigrum, Erica tetralix* and *Galium*

Table 2.3

The MAM mixed-effects model relating species richness (number of species m^{-2}) with respect to moor and (a) elapsed time since burning, and (b) vegetation height. The results are presented for (i) the entire temporal dataset, and (b) the dataset with the old stands excluded. The fitted estimate, the standard error (SE), Z value and *P*-value is provided for each term. The AIC statistic is shown for both the MAM and the null model along with the % reduction (%Δ). Significance of each term is coded: ns= >0.10; + = P<0.10, **=P<0.01, ***=P<0.001. The intercept in both models is Bamford Moor.

(i) Entire dataset

(a) Elapsed time since burning (Et) - AIC: Null model=723; MAM=587; Δ=18.7%

Parameter		Moor						Interaction term		
	Intercept	Broomhead	Howden	Midhope	Snailsden	Et	Broomhead x Et	Howden x Et	Midhope x Et	Snailsden x Et
Estimate	2.410	-0.894	-0.226	-0.865	-0.681	-0.037	0.029	0.014	0.029	0.021
SE	0.050	0.080	0.072	0.083	0.089	0.004	0.005	0.005	0.006	0.006
Z	48.370	-11.110	-3.130	-10.450	-7.660	-9.360	5.710	2.630	5.330	3.650
P	<0.001***	<0.001***	<0.002**	<0.001***	<0.001***	<0.001***	<0.001***	<0.009**	<0.001***	<0.001***

(b) Vegetation height (Ht) - AIC: Null model=832; MAM=602; Δ=27.6%

Parameter		Moor						**Interaction term**		
	Intercept	Broomhead	Howden	Midhope	Snailsden	Ht	Broomhead xHt	Howden x Ht	Midhope x Ht	Snailsden x Ht
Estimate	2.330	-0.884	-0.177	-0.833	-0.585	-0.020	0.019	0.005	0.018	0.013
SE	0.038	0.076	0.056	0.086	0.073	0.002	0.003	0.003	0.003	0.003
Z	61.000	-11.560	-3.190	-9.740	-7.990	-9.530	7.000	1.930	6.140	4.350
P	<0.001***	<0.001***	<0.001**	<0.001***	<0.001***	<0.001***	<0.001***	<0.009**	<0.01^{+}	<0.001***

(ii) Old patches excluded

(a) Elapsed time since burning (Et) - AIC: Null model=529; MAM=513; Δ=3.0%

Parameter		Moor			
	Intercept	Broomhead	Howden	Midhope	Snailsden
Estimate	2.11906	-0.66111	-0.15475	-0.66336	-0.5282
SE	0.03175	0.05516	0.04642	0.05499	0.05341
Z	66.75	-11.99	-3.33	-12.06	-9.89
P	<0.001***	<0.001***	<0.001**	<0.001***	<0.001***

(b) Vegetation height (Ht) - AIC: Null model=544; MAM=521; Δ=4.2%

Parameter		Moor						Interaction term		
	Intercept	Broomhead	Howden	Midhope	Snailsden	Ht	Broomhead x Ht	Howden x Ht	Midhope x Ht	Snailsden x Ht
Estimate	2.195	-0.951	-0.174	-0.783	-0.542	-0.007	0.016	0.002	0.009	0.005
SE	0.049	0.098	0.076	0.111	0.092	0.004	0.005	0.005	0.005	0.005
Z	45.17	-9.7	-2.28	-7.08	-5.88	-2.12	3.51	0.47	1.77	1
P	<0.001***	<0.001***	<0.05*	<0.001***	<0.001***	<0.05*	<0.001***	ns	<0.01^{+}	ns

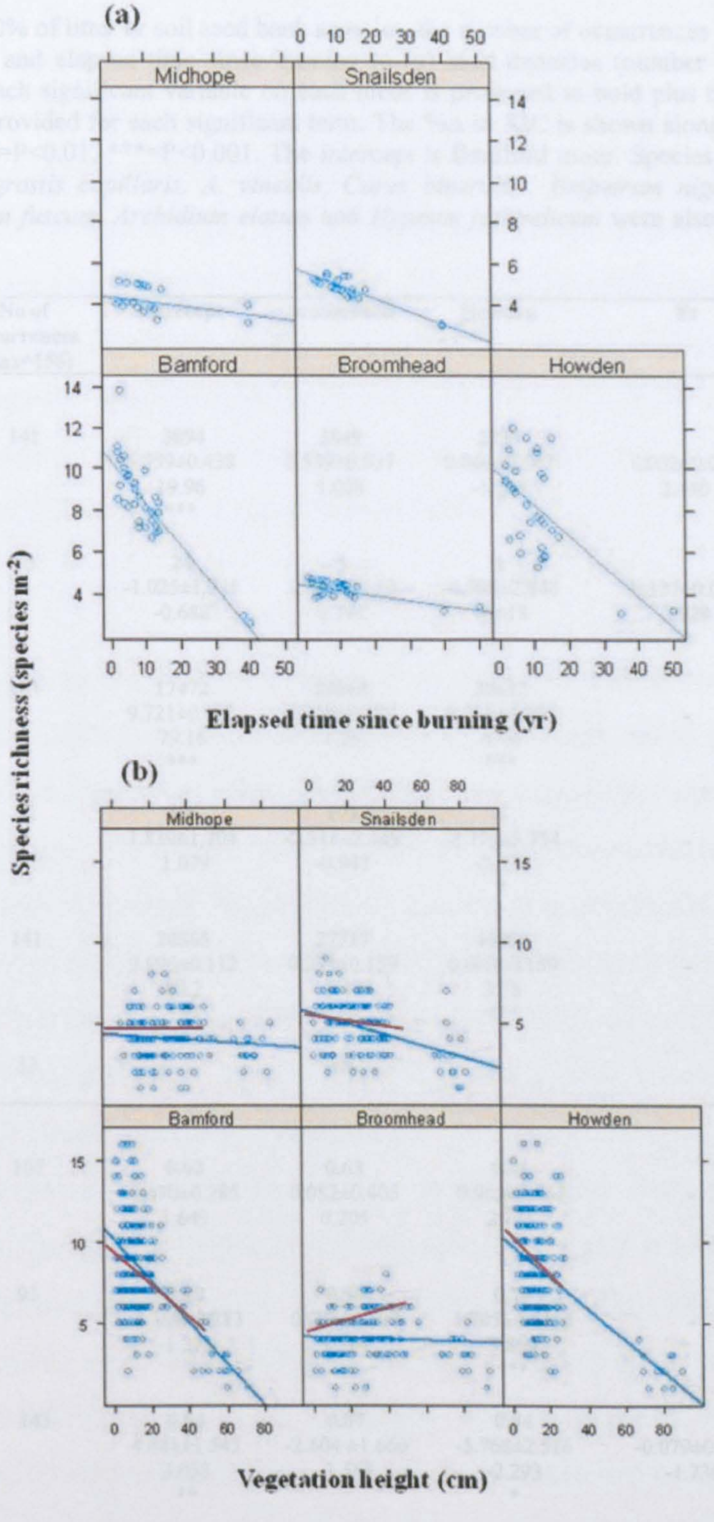

Fig 2.1. Response of species richness to (a) elapsed time since burning (yr), and (b) vegetation height (cm) at five moors in the Peak District, Derbyshire. Fitted lines were derived from mixed-effects models (Table 2.4); (b) blue lines are of the entire dataset, the brown lines have the older stands removed.

Table 2.5.

The species detected in more than 10% of litter or soil seed bank samples, the number of occurrences out of 156 and the minimum adequate mixed-effects model relating moor and elapsed time since burning to (a) seed densities (number of seedlings m^{-2}), and (b) bryophyte presence. The arithmetic mean of each significant variable on each moor is presented in bold plus the fitted model estimate, the standard error (SE), Z value and *P*-value is provided for each significant term. The %Δ in AIC is shown along with the significance of each term is coded as: ns= $P>0.05$; *=$P<0.05$;**=$P<0.01$, ***=$P<0.001$. The intercept is Bamford moor. Species detected in <10% of samples of both litter and soil seed banks were *Agrostis capillaris, A. vinealis, Carex binervis, , Empetrum nigrum, Erica tetralix, Galium saxatile, Polytricum formosum* and *Sphagnum fuscum*. *Archidium elatum* and *Hypnum jutlandicum* were also present at low occurrence in the soil seed bank.

Fraction	Species	No of occurrences (max=156)	Intercept	Broomhead	Howden	Et	Broomhead x Et	Howden x Et
(a)								
Litter	*Calluna vulgaris*	141	**3094**	**3849**	**2239**			
	Mean±SE		6.559±0.438	0.539±0.537	0.060±0.547	0.032±0.013	-	-
	Z		19.96	1.003	-1.104	2.440		
	ΔAIC= <0.1%		***			*		
	Juncus effusus	33	**24**	**3**	**1**			
	Mean±SE		-1.025±1.241	1.241±3.150	-4.598±2.840	0.137±0.068	-0.375±0.298	-0.090±0.097
	Z		-0.688	0.394	-1.618	2.029	-1.262	-0.935
	ΔAIC= <0.4%					*		
Soil	*Calluna vulgaris*	141	**17472**	**23868**	**38617**			
	Mean±SE		9.721±0.123	0.219±0.174	0.718±0.207	-	-	-
	Z		79.16	1.26	4.14			
	ΔAIC= 1.3%		***		***			
	Juncus effusus	22	**486**	**179**	**5**			
	Mean±SE		1.839±1.704	-2.318±2.449	-8.775±3.754	-	-	-
	Z		1.079	-0.947	-2.338			
	ΔAIC=<0.1%				*			
Total	*Calluna vulgaris*	141	**20565**	**27717**	**40855**			
	Mean±SE		9.896±0.112	0.234±0.159	0.600±0.159	-	-	-
	Z		88.2	1.47	3.78			
	ΔAIC=<0.1%		***		***			
	Juncus effusus	33		ns			-	-
(b)								
Litter	*Campylopus introflexus*	107	**0.62**	**0.63**	**0.81**			
	Mean±SE		0.470±0.285	0.082±0.405	0.965±0.453	-	-	-
	Z		1.649	0.205	2.131			
	ΔAIC=0.8%				*			
	Campylopus pyriformis	91	**0.40**	**0.56**	**0.79**			
	Mean±SE		-0.389±0.283	0.623±0.397	1.705±0.4418	-	-	-
	Z		-1.378	1.564	3.860			
	ΔAIC=5.9%				***			
	Pohlia nutans	143	**0.94**	**0.87**	**0.94**			
	Mean±SE		4.681±1.543	-2.604 ±1.660	-5.768±2.516	-0.079±0.046	0.069±0.050	0.572±0.344
	Z		3.033	-1.568	-2.293	-1.736	1.370	1.661
	ΔAIC=3.3%		**		*			
	Archidium elatum	51	-	-	-	-	-	-
	Hypnum jutlandicum	42	**0.44**	**0.28**	**0.11**			
	Mean±SE		-0.249±0.322	-0.926±0.484	-1.905±0.576	-	-	-
	Z		-0.774	-1.913	-3.309			
	ΔAIC=4.0%				***			
Soil	*Campylopus pyriformis*	142	**0.94**	**0.98**	**0.81**			
	Mean±SE		3.744± 0.723	1.479±1.202	-1.892±0.714	-0.046±0.016	-	-
	Z		5.178	1.230	-1.664	-2.876		
	ΔAIC=7.6%		***			**		
	Pohlia nutans	50	Ns	-	-	-	-	-

saxatile were detected in a few samples (n<16). Of the bryophytes five species were detected in either the litter or soil that had a great enough frequency to analyze, these are: *Archidium elatum*, *Campylopus introflexus*, *Campylopus pyriformis*, *Hypnum jutlandicum*, *Pohlia nutans* ; *Polytricum formosum* and *Sphagnum fuscum* were detected at low frequency.

All species that showed a significant relationship relative to the null model indicated significant differences between the different moors. Where an additive effect between Moors and prescribed burning (common slope) was detected, one was positive (*Calluna* in litter) and the other negative (*Campylopus pyriformis* in the soil). Where an interaction was detected (*Juncus effusus* and *Pohlia nutans* in litter), the slopes with respect to elapsed time since burning were different on the three moors (Table 2.5).

2.4. Discussion

There is an increasing requirement to develop evidence-based conservation policies (Pullin & Knight, 2009), and this is especially important when management has to encompass multiple, and possibly conflicting, conservation requirements (Marrs *et al.*, 2007). Here, we investigate just one aspect, the conservation of plant species richness in upland moorland, where there has been considerable policy debate on future management objectives (Tucker, 2003). Our research was based within the Peak District in northern England, a National Park with 22 million day visitors per year in 1998 (Peak District National Park, 1998), and where the vegetation is acknowledged as being severely damaged by industrial airborne pollutants from the surrounding industrial conurbations. Our study is the first regional, multi-site chronosequence study of its kind on British moorlands and it aimed to provide (a) a first snapshot assessment of the current vegetation, (b) an assessment of the restoration potential from the seedbank, and (c) a view on the impact of prescribed burning on species richness. The intention was to provide evidence to inform conservation policy decisions in these areas where there are conflicting demands for current management, conservation of species or ecosystem services (Marrs *et al.*, 2007).

2.4.1. Restoration potential of the vegetation

Unfortunately, there is no historic monitoring data from these moors against which restoration progress can be judged, however, this study was designed to be statistically rigorous and thus provide a baseline for future comparisons. In order to assess the state of the vegetation this study used a comparison with the NVC of the UK (Rodwell, 1991, 1992). This is a crude method but allowed an assessment of each sampled quadrat relative to the NVC class that it most matched. The alternative of choosing a "standard" NVC class for all to aspire to was rejected because of the variations in stakeholder requirements for the land management. These results showed that there is a considerable β-diversity across these five moors, with 28 different NVC sub-communities and communities, and this was perhaps surprising given the apparent lack of obvious heterogeneity in the vegetation. The majority of the quadrats were subsumed in seven NVC classes, and these reflect communities ranging from heath and upland grassland on the drier areas (H1, H9, H11, H12, U2) through to mire communities (M19, M20) in the wetter parts. The majority of these communities contain many of the same dominant species, albeit in differing proportions (Table 3); (Rodwell, 1991, 1992) and the frequencies of sub-dominants are important in differentiating between them. However, as demonstrated here, many of the sub-dominant species that might be expected to occur in the Peak District region were missing, and this of course is one of the reasons for the variable and often poor goodness-of-fit detected.

Thus the communities detected represent a complex mosaic of moorland vegetation occurring on these peat soils that can be classified at present along a heath-mire community gradient. However, all communities are missing member species with respect to target community lists, a result that is in agreement with historic studies suggesting that the Moors are depauperate as a result of past pollution (Tallis, 1964, 1998; Ferguson & Lee, 1983), and damaging land management practice (Anderson & Yalden, 1981). Indeed, burning has been implicated in the reduction or extirpation of many rare species (McVean & Ratcliffe, 1962; Rodwell, 1991; Ratcliffe, 2002; Preston *et al.*, 2000), and the perpetuation of the dominant species *Calluna vulgaris* (McVean & Ratcliffe, 1962). In these accounts, however, it is often unclear whether the response observed was due to prescribed burning rather than wildfire, where a greater fire intensity, fire severity and ecosystem response *sensu* (Keeley, 2002) would be expected. Moreover, at least for some species, other factors were also noted as important, especially drainage and overgrazing. Irrespective, none of the species identified in this work as being damaged by fire were detected in this survey! Of course it could be argued that burning has already removed them, but from a restoration perspective they are absent. There is no doubt, however, that *Calluna* responds well to prescribed burning.

This implication of these findings is that to restore these moorlands in keeping with EU legislation (Anon, 2006), the species complement should be enhanced with those species that are currently missing. The results from the study of propagule distribution and abundance suggests that the propagule banks have a low species richness and abundance; *Calluna* is the only species that has a seed bank comparable with other upland moors (Pakeman & Hay, 1993), all other species are lower than other moorlands albeit of lower altitude (Ghorbani *et al.*, 2006, 2007). Essentially, the species missing from the vegetation will not be restored from the extant propagule banks.

2.4.2. Impact of prescribed burning on species composition

With respect to our hypotheses on the impact of prescribed burning, the analyses of the entire dataset rejected Hypothesis 1 that prescribed burning has no effect on plant species richness. The mixed-effects models showed that there were (a) differences between the five moorlands, and (b) negative slopes with respect to both elapsed time since burning and vegetation height, The differences between moors is important because it implies that there is variation in species richness between management units, which is probably brought about through a combination of landscape factors and both past and present land-management history. The relationships with elapsed time since burning and vegetation height also differed between moors. Taken together these results suggest that it would be unwise to develop "one-size-fits–all" management prescriptions. These negative slopes also suggest that prescribed burning enhances species richness in the immediate post-burn period, with a subsequent decline as the vegetation ages and height increases. Hence, Hypothesis 2 was accepted, that there was a positive effect on plant species richness, although this was relatively short term (15-20 years; height <50cm) and there was a decline in the oldest burned patches. Vegetation height may be a more useful variable for moorland managers to use to predict the species richness of their moors as it is easy to measure under field conditions. There was no evidence to support Hypothesis 3 that prescribed burning has a negative effect on plant species richness.

Table 2. 4.
The MAM mixed-effects model relating species richness (number of species m^{-2}) with respect to moor and (a) elapsed time since burning, and (b) vegetation height. The results are presented for (i) the entire temporal dataset, and (b) the dataset with the old stands excluded. The fitted estimate, the standard error (SE), Z value and *P*-value is provided for each term. The AIC statistic is shown for both the MAM and the null model along with the % reduction (%Δ). Significance of each term is coded: ns= >0.10; + = P<0.10, **=P<0.01, ***=P<0.001. The intercept in both models is Bamford Moor.

(i) Entire dataset

(a) Elapsed time since burning (Et) - AIC: Null model=723; MAM=587; Δ=18.7%

Parameter	Moor						Interaction term			
	Intercept	Broomhead	Howden	Midhope	Snailsden	Et	Broomhead x Et	Howden x Et	Midhope x Et	Snailsden x Et
Estimate	2.410	-0.894	-0.226	-0.865	-0.681	-0.037	0.029	0.014	0.029	0.021
SE	0.050	0.080	0.072	0.083	0.089	0.004	0.005	0.005	0.006	0.006
Z	48.370	-11.110	-3.130	-10.450	-7.660	-9.360	5.710	2.630	5.330	3.650
P	<0.001***	<0.001***	<0.002**	<0.001***	<0.001***	<0.001***	<0.001***	<0.009**	<0.001***	<0.001***

(b) Vegetation height (Ht) - AIC: Null model=832; MAM=602; Δ=27.6%

Parameter	Moor						**Interaction term**			
	Intercept	Broomhead	Howden	Midhope	Snailsden	Ht	Broomhead xHt	Howden x Ht	Midhope x Ht	Snailsden x Ht
Estimate	2.330	-0.884	-0.177	-0.833	-0.585	-0.020	0.019	0.005	0.018	0.013
SE	0.038	0.076	0.056	0.086	0.073	0.002	0.003	0.003	0.003	0.003
Z	61.000	-11.560	-3.190	-9.740	-7.990	-9.530	7.000	1.930	6.140	4.350
P	<0.001***	<0.001***	<0.001**	<0.001***	<0.001***	<0.001***	<0.001***	<0.009**	<0.01+	<0.001***

(ii) Old patches excluded

(a) Elapsed time since burning (Et) - AIC: Null model=529; MAM=513; Δ=3.0%

Parameter	Moor				
	Intercept	Broomhead	Howden	Midhope	Snailsden
Estimate	2.11906	-0.66111	-0.15475	-0.66336	-0.5282
SE	0.03175	0.05516	0.04642	0.05499	0.05341
Z	66.75	-11.99	-3.33	-12.06	-9.89
P	<0.001***	<0.001***	<0.001**	<0.001***	<0.001***

(b) Vegetation height (Ht) - AIC: Null model=544; MAM=521; Δ=4.2%

Parameter	Moor						Interaction term			
	Intercept	Broomhead	Howden	Midhope	Snailsden	Ht	Broomhead x Ht	Howden x Ht	Midhope x Ht	Snailsden x Ht
Estimate	2.195	-0.951	-0.174	-0.783	-0.542	-0.007	0.016	0.002	0.009	0.005
SE	0.049	0.098	0.076	0.111	0.092	0.004	0.005	0.005	0.005	0.005
Z	45.17	-9.7	-2.28	-7.08	-5.88	-2.12	3.51	0.47	1.77	1
P	<0.001***	<0.001***	<0.05*	<0.001***	<0.001***	<0.05*	<0.001***	ns	<0.01+	ns

There are limitations to this work; the space-for-time substitution approach can always be criticised because it assumes that vegetation in all samples places responds in the same way and at the same rate. However, in this study we used a multi-site and multi-patch approach, where the burned patches on each moor were selected randomly from a large pool of potential patches for each age on every sampling occasion. Thus the chronosequence was sampled without bias. The estimation of patch age was possible here because the five moorlands have good historical management records. However, it is possible that there is inaccuracy in age estimation, which would of course affect the statistical analysis. In order to ensure that the conclusions from the temporal study were valid, the analyses were corroborated using vegetation height as the independent variable. Height was highly correlated with age ($r=0.80$, $P<0.0001$, $df=1008$), but was measured independently at each quadrat. Moreover, the analyses of the reduced dataset where these older patches were excluded provided confirmatory results. In the analysis relating species richness to elapsed time, there were only significant differences between moors and no significant slope with elapsed time since burning. Over the restricted height range, negative slopes were found in four of the five moors (two significant) and only one showed a significant positive response.

To some extent the conclusions on the relationship between species richness and post-fire succession are not surprising. In a similar chronosequence in Scotland on a single site (Dinnet Moor) a similar reduction in species richness with time after burning was shown (Hobbs & Gimingham, 1984). In what these authors term "species-rich" heathland (17-29 species in the 0-25 year period), they reported a reduction in grass, forb and lichen growth, and notably little grass and forb regrowth in older stands. In a similar, single-site study of burning on lichen diversity, the immediate effect of fire was to reduce lichen diversity; however, it recovered within 20 years, and thereafter declined (Davies & Legg, 2008). Stands more than 25 years old generally had a lower diversity than stands 10-15 years old. These authors concluded that fire was an important factor in maintaining lichen diversity in these managed Scottish heathlands. Unfortunately, lichen presence on the Peak District moors is low so we could not test this hypothesis.

In the only replicated experiment of burning on moorland vegetation, (Rawes & Hobbs, 1979) concluded that "light grazing by sheep and no burning, is likely to be an acceptable management strategy in the interests of conservation". However, this conclusion cannot be fully justified from their results. They showed an *Eriophorum vaginatum* peak before the *Calluna* one, suggesting that burning was important in maintaining the graminoid:*Calluna* balance, and many species, including *Sphagnum* species, were at their maximum abundance in a 10-year rotation burn treatment compared to a 20-year rotation.

However, our results in species-poor moorland, recovering from the effects of pollution, show a similar response to the more species-rich moorlands. Taken together, the results suggest that species richness is enhanced by prescribed burning and where burning is not carried out for over 30 years then species richness, and hence conservation value, is reduced. This is consistent with the Intermediate-Disturbance hypothesis of (Connell, 1978), where disturbance (here using prescribed fire) maintains greater plant species diversity than when left undisturbed. It is probable that competition for light as the vegetation develops reduces the species diversity (Hautier *et al.*, 2009).

Our results confirm that prescribed moorland burning is a key disturbance factor that, at present, is maintaining current plant species diversity. Moreover, the balance of evidence presented here suggests that a lack of burning and specifically a no-burn policy will reduce this diversity. Moreover, to restore these moorlands to a target NVC community will require addition of propagules, but here the methodology needs to be tested. These moorlands are currently composed of many plant species that have evolved to respond to disturbance, but this disturbance needs to be implemented within a cultural landscape that is used for conservation, shooting interests, recreation, carbon sequestration and water harvesting. Here, we only consider one of these uses, the conservation of the current plant species diversity. Nevertheless, we hope that our conclusions can be used to develop evidence-based conservation practice (Stewart *et al.*, 2005; Pullin & Knight, 2000; Driscoll *et al.*, 2010).

Chapter 3

Factors affecting moorland plant communities and component species in relation to prescribed burning in the Peak District, England

3. Factors affecting moorland plant communities and component species in relation to prescribed burning in the Peak District, England

3.1. Introduction

The upland moors in Great Britain have a high conservation value of international significance (Ratcliffe & Thompson, 1988) as they contain six communities that are virtually confined to the UK (Thompson *et al.*, 1995). They are cultural landscapes, created and maintained by anthropogenic activity, which prevents succession to woodland and maintains a very nutrient-poor ecosystem (Gimingham, 1972; Marrs, 1993). The vegetation is dominated by dwarf shrubs, mainly *Calluna vulgaris* (Gimingham, 1972) and currently they are managed for sheep grazing and sporting interests (shooting red grouse, *Lagopus lagopus scoticus* (L.). Nomenclature follows (Stace, 1997) for higher plants, (Atherton *et al.*, 2010) for bryophytes, and (Dobson, 2000) for lichens). Moorlands also provide a habitat for other bird species such as curlew *Numenius arquata* (L.), golden plover *Pluvialis apricaria* (L.), black grouse *Tetrao tetrix* (L.), merlin (*Falco columbarius L.)*, hen harrier (*Circus cyaneus L.)*, short-eared owl *Asio flammeus* (Pontoppidan) and ring ouzel *Turdus torquatus* (L.) (Robson, 1998; Whittingham *et al.*, 2000); Thompson *et al.*, 1997). The perpetuation of these moorlands is important in terms of both the local economy and biodiversity. Grouse shooting alone is worth 12% of the total UK shooting economy with an estimated £120 million spent in the uplands during good years (PACEC, 2006).

Most moorland is managed using prescribed burning. Burning has been used for hundreds, perhaps thousands of years (Pearsall, 1950; Birks, 1988; Simmons, 2003), although it has increased in frequency over the last 150 years as a result of grouse management (Bonn *et al.*, 2009). The usual management practice is burn on a rotation to produce a mosaic of stands in different stages of the burn-recovery cycle. The recovering *Calluna* produces forage for red grouse and older, taller vegetation provides cover (Miller, 1980). The aim of the prescribed burning is to remove woody plant growth and any old degenerate plants, leaving charred stems. The burnt stems resprout to produce new growth, although regeneration reduces with plant age, and falls off sharply after ca. 15 years of age (Lovat, 1911; Miller & Miles, 1970). Where resprouting is slow, plant species may need to regenerate from seed, which is either buried in the soil or disperses from outside the burn patch (Gimingham, 1994).

Approximately, 65% of British upland moors are managed using this mosaic approach for the benefit of red grouse (Sotherton *et al.*, 2009).

Recently the use of burning has been the subject of review (Tucker, 2003) and the consequence of this review has been substantive debate on whether prescribed burning should be continued. The reason for this debate is that moorlands must now contribute to an increasing range of conservation objectives, ranging from the conservation of species and communities through to the provision of ecosystem services (carbon sequestration, water provision and recreation) (MEA, 2005; Marrs, *et al.*, 2007). Given the critical, and in some cases controversial, discussions surrounding these issues, it is surprising given our increasing requirement to have evidence-based conservation management (Pullin & Knight, 2009; Pullin, *et al.*, 2009) that so little is known about the community dynamics of post-fire succession. Some evidence is available from single sites in Scotland for higher plants (Hobbs & Gimingham, 1984) and for lichens (Davies & Legg, 2008). However, there is a lack of evidence on plant communities in areas on England, where the debate over prescribed burning is at its most contentious. The one experiment where different rotations were tested provided equivocal results (Rawes & Hobb, 1979); the overall conclusions suggested cessation of burning was "likely to be an acceptable management in the interests of conservation", but this conclusion could not be fully justified from their results. They showed an *Eriophorum vaginatum* peak before the *Calluna* one suggesting that burning was important in maintaining the graminoid:*Calluna* balance, and many species, including *Sphagnum* species, were at their maximum abundance in a 10-year rotation burn treatment compared to a 20-year one. This lack of predictive reliable information to inform conservation policy has been confirmed in a systematic review, which assessed the impact of burning as a conservation intervention on heaths and bogs: the outcome was that evidence was insufficient to generate robust management recommendations (Stewart, *et al.*, 2005).

Here, we assess the impact of prescribed burning in a multi-site study, where we use a space-for-time substitution or chronosequence approach to assess changes in plant species community compositions on five moorlands. Multivariate analyses were used to assess community composition and this was related to both the moorland site and a range of environmental factors (ter Braak & Šmilauer, 2005), that included both elapsed time since burning and vegetation height. The aim of this study was to determine which environmental factors were significantly correlated with species community composition. Thereafter, the realized niche of each species (Austin, *et al.*, 1990; Lawesson & Oksanen, 2002; Smart *et al.*, 2010) was calculated with respect to both elapsed time since burning and vegetation. Here the Huisman, Olff and Fresco (HOF) modelling approach (Huisman, *et al.*, 1993) was used to calculate the response curves. Although other approaches including Gaussian response curves (Lawesson & Oksanen, 2002), Beta response functions and Generalized Additive Models (Oksanen & Minchin, 2002) can also be used, their applicability has been previously criticized, and HOF modelling has been suggested as an adequate substitution (Lawesson & Oksanen, 2002). We hypothesise that the intermediate disturbance caused by prescribed burning causes an increase in species richness. Here, we test this hypothesis by analysing the response of plant communities with respect to a range of environmental factors, and through inspection of the individual species response curves describing their realized niche space. Moreover, there is a need to identify the environmental factors that influence community composition with respect to prescribed fire. Ultimately, the aim was to derive simple measures that could help moorland managers to maintain maximal species diversity on their land.

3.2. Methods

3.2.1. Study areas and sampling protocol

A space-for-time substitution study was carried out on five moorlands (Bamford, Broomhead, Howden, Midhope, and Snailsden Moors) in the North Peak ESA (Environmentally Sensitive Area), within the Peak District National Park, UK over a three-year period (Table 1). All of the burned patches on each Moor were mapped using aerial photography taken in September 2005 and cross-referenced with land burning management maps provided by the land managers. This provided a series of burned patches of known age (elapsed time since burning) between 2-16 years; older patches were also identified where no burning had been carried out for at least 35 years. There were 79, 249,103, 271, and 252 available burn patches on the five moors respectively. For the vegetation 10 different burns were selected using an age-stratified random sampling procedure from each Moor in each of the three survey years; the older patches were sampled in 2007 and 2008 (Table 3.1). Once selected, the geo-referenced outlines of each of the sampled patches were digitized from aerial photographs, and their areas calculated within (ArcGIS, 2009). The number of $1m^2$ quadrats available within each burn patch was counted and a random selection made for field sampling (vegetation; n=10 in 2006 and 2008, n=4 in 2007, total n=1010).

Table 3.1 Details of the five study moorlands in the Peak District National Park, England where the post-fire vegetation succession was surveyed.

Moorland site	Longitude & Latitude	British National Grid squares digitized per Moor	Elevation range (m)	Age range (yrs)	Estimated age of older patches (yr)	Sampling years
Bamford	Latitude 53°21'N, Longitude1°40'W	SK 1993, 1994, 2093, 2094;	300-420	3-14	38	2006, 2007, 2008
Broomhead	Latitude 53°27'N, Longitude1°38'W	SK 2395, 2394, 2295 2294;	300-460	2-15	40	2007, 2008
Howden	Latitude 53°28'N, Longitude1°42'W	SK 2184, 2185, 2284, 2285;	272-540	2-15	50	2006, 2007, 2008
Midhope	Latitude 53°29'N, Longitude1°40'W	SK2198, 2197, 2099, 2098, 2097, 1999, 1998, 1997;	270-480	3-15	40	2007, 2008
Snailsden	Latitude 53°30'N, Longitude1°44'W	SE1503, 1501, 1404, 1401, 1400, 1304.	350-470	3-16	50	2007, 2008

Table 3.2 Environmental variables measured or derived for the study at each of the five moors in the Peak District National Park, England. The allocation of each variable to the environmental dataset and the transformation used is presented, along those variables in each set selected after forward selection. The ΔAIC criterion for each within-set model is presents as a % reduction relative to the null model (AIC=3239).

Variable name	Description	Source	Transformation	Environmental set used in Variation partitioning and Forward Selection	Variables significant after within-set forward selection	ΔAIC (%) of selected model relative to the null model
Moor	Moor Name	Ordinance Survey Maps	-	Site	✓	Site=5.2
Easting	National Grid (km)	Ordinance Survey Maps	Standardized (mean=0, s^2=1)	Site	✓	
Northing	National Grid (km)	Ordinance Survey Maps	Standardized (mean=0, s^2=1)	Site	✓	
Elevation	Height above mean sea level (m)	Ordinance Survey Maps	$\log_e(x+1)$	Site	✓	
Aspect	Measured compass bearing (°)	Magnetic compass(corrected for magnetic anomaly)	Arcsin(sqrt(x/100))	Site		
Faspect	Functional transformation (F) of Aspect (a, degrees, the estimated site-wise mean), $F=\lvert -\sin(a/2)\rvert$	Derived from Aspect	-	Site		
Slope	Mean gradient of survey site(degrees)	Clinometer	Arcsin(sqrt(x/100))	Site	✓	
Burnt *Calluna* bush	Cover (%)	Estimated on site in 1m² quadrats	Arcsin(sqrt(x/100))	Biotic	✓	Biotic=3.7
Burnt *Calluna* stick	Cover (%)	Estimated on site in 1m² quadrats	Arcsin(sqrt(x/100))	Biotic	✓	
Othelitt	Cover (%) of other litter	Estimated on site in 1m² quadrats	Arcsin(sqrt(x/100))	Biotic	✓	
Baregrou	Cover (%) of bare ground	Estimated on site in 1m² quadats	Arcsin(sqrt(x/100))	Biotic		
Animexcr	Cover (%) of animal excrement	Estimated on site in 1m² quadrats	Arcsin(sqrt(x/100))	Biotic	✓	
pH	$pH = -\log_{10}[H^+]$	Measured in laboratory (Allen 1989)	-	Physical	✓	Physical=4.0
LOI	Amount of organic matter present in the soil	Measured in laboratory (Ball 1964)	Arcsin(sqrt(x/100))	Physical		
Rock	Cover (%) of Rock	Estimated on site in 1m² quadrats	Arcsin(sqrt(x/100))	Physical		
Open water	Cover (%) of open water	Estimated on site in 1m² quadrats	Arcsin(sqrt(x/100))	Physical		
Biomass	Sampled at each quadrat (g m^{-2})	Measured in laboratory; data available as fresh and dry weights	$\log_e(x+1)$	Production		Production=6.2
Vegetation moisture content	Sampled at each quadrat	Measured in laboratory; data available as mass and %	Mass: $\log_e(x+1)$; %= Arcsin(sqrt(x/100))	Production	✓	
Elapsed time since last burn (Et)	Elapsed time (year) since burning	Estate management		Production	✓	
PD	Measurement (cm) of vegetation height within the quadrat pressure, using pressure disk. Pressure of disk on vegetation $P = \text{Force/Area} - (m \times g)/(\pi \times r^2) = 27.76$ Pa	Measured on site with a pressure disk – diameter= 0.3 m, mass 0.2kg	sqrt(x+0.5)	Production	✓	
Sampling year	2006,2007,2008	Year observations were made	-	Year	✓	Covariable=3.2

Table 3.3 The relative contributions of four sets of environmental variables in explaining species composition in the Variation Partitioning. The four sets use the significant variables detailed in Table 2; sets are coded - S=Site, B=Biotic, Ph=Physical, Pr=Production. Pseudo-F was derived from a randomization test with 999 permutations stratified within Moor; all are significant (**=P<0.01; ***=P<0.001).The vertical lines show the constrained variable on the left and the covariables on the right.

Variable set	Df	Adjusted r^2 (%)	Pseudo-F
S+B+Ph+Pr	12	13.2	17.4***
S	4	5.0	14.2***
B	4	8.1	2.4**
Pr	3	7.2	27.0***
Ph	1	0.9	9.7***
S|B+Pr+Ph	4	2.4	8.2***
B|S+Pr+Ph	4	1.7	15.6***
Pr|S+B+Ph	3	2.0	55.3***
Ph|S+B+Pr	1	0.6	7.6***

Table 3.4 Upper tolerance limits of species with a fitted HOF Model type IV and V at each of the five moors in the Peak District, National Park, England (nc= upper tolerance not calculated).

Species	Vegetation Height (cm)	Elapsed time (year)
Agrostis capillaris	14.6	15.4
Carex pilulifera	nc	21.6
Cladonia chlorophea	38.5	21.7
Campylopus introflexus	20.2	20.3
Cladonia squamosa	11.6	15.6
Deschampsia flexuosa	26.4	11.0
Dicranum scoparium	20.0	15.1
Erica tetralix	nc	23.3
Eriophorum angustifolium	28.4	Nc
Eriophorum vaginatum	nc	36.7
Empetrum nigrum	46.9	21.8
Hypnum jutlandicum	37.2	20.3
Nardus stricta	13.2	Nc
Polytrichum commune	40.8	Nc

3.2.2. Vegetation survey

On each of the five moors the patches and then the quadrat positions were located using GPS (eTrex Venture® HC). The cover of all higher plants, bryophytes and lichens were recorded, as well as an assessment of the *Calluna* response to burning in two cover categories, either "Bush" or "Stick". These two categories represent management description of burning response. The "Bush" category reflects plants that recover normally from burning.), and the "Stick" category reflects older *Calluna* that has only been scorched, yet remains alive within a younger developing sward. A range of environmental measurements were also made (Table 3.2) including: aspect (°), slope (°) and vegetation height (cm). Vegetation height was measured by placing a disc (diameter area=0.30m, mass=0.200kg) in the centre of every quadrat and vegetation height (cm) measured as distance from the disc to the ground (Stewart, *et al.* 2001). A vegetation sample was harvested from the central 0.25 m^2 sorted into five fractions (dwarf shrubs, graminoids, bryophytes, litter, animal excrement), dried at 80°C and weighed. On four moors, all samples had a peat depth of at least 50 cm, the exception was Midhope where some quadrats were on shallower peat (20-30cm =4.3%; 30-40cm=14.2%, 40-50cm=3.7%; >50cm = 77.8%).

3.2.3. Data analysis

Multivariate analysis and HOF-modelling were performed using the 'vegan' and 'gravy' packages respectively (Oksanen, 2004, 2005) within the R statistical environment (R Development Core Team, 2010).

(a) Multivariate analysis

All multivariate analyses used transformed species cover data ($\log_e(y+1)$). Initially, these data were analysed using Detrended Correspondence Analysis (DCA) using the 'vegan' function 'decorana'. The DCA produced eigenvalues of 0.307, 0.136, 0.127 and 0.1263, and axis lengths of 3.396, 2.583, 2.525 and 3.257 for the first four axes; as the gradient length for axis 1 was greater than three, the unimodal model Canonical Correspondence Analysis (CCA) was adopted for direct gradient analysis (ter Braak & Šmilauer, 2005). In order to select the most significant environmental variables for use in both direct gradient analysis and variation partitioning, a two-phased approach was used (Corney *et al.*, 2006). First, the environmental variables were allocated into five sets, namely {Site}, {Biotic}, {Physical}, {Production} and {Sampling Year} (Table 3.2). Within each set the most significant variables were selected using CCA ('cca' function in 'vegan') using the forward selection procedure and the AIC statistic as the selection criterion. In this analysis, year of sampling was included as a covariable. The selected variables within each set are presented in Table 3.2 along with the AIC reduction for each model. Second, CCA was run again as above, but here using all variables selected in stage 1 to relate species composition to environmental factors. The significance of the final CCA model was assessed using a randomization test with 999 permutations restricted within Moor and Year. The distribution of each moor was plotted on the CCA quadrat plot as a bivariate ellipse (standard deviation + 95% confidence limits) and the ellipse area calculated using 'vegan' function 'ordiellipse'. Variation partitioning (Peres-Neto, *et al.*, 2006) was then carried out using the 'varpart' function in vegan (Oksanen, 2005) to assess the relative contribution of each set in explaining the variation in species composition. The variation partitioning was carried out using those significant variables within the four environmental sets identified in Table 2. Significance of each of the testable fractions of the VP analysis was done using Redundancy Analysis (RDA) with a randomization test and 999 permutations restricted within Moor and Year.

(b) Modelling post-burning species responses

There were several variables that could be used as the gradient in this analysis. The most obvious one would be elapsed time since burning, although vegetation height or vegetation biomass could also be used as surrogates. Here, elapsed times since burning and vegetation height were selected. The reasons are for this were, first, whilst every attempt was made to ensure that estimates of elapsed time since burning were accurate using historical information, there is some uncertainty, especially for the oldest burned patches. Any inaccuracy here could skew model fitting. Second, vegetation height was the most significant variable determined in the forward selection procedure discussed above, and was more or less co-incident with elapsed time since burning in the ordination diagram. Third, vegetation height is easy to measure and hence its use could easily be translated into practical use.

Thus, the HOF modelling procedures was used to fit species cover to vegetation height. This procedure fits five models in an hierarchical sequence: Model Type 1 has no significant response (null model), Type II has an increasing or decreasing response up to the maximum potential cover, Type III is similar to Type II but has an asymptote below the maximum possible, Type IV is an unimodal response and Type V is similar to Type IV but accommodates a skewed response (Huisman *et al.*, 1993). Here the HOF models were modified with a Poisson error distribution and the AIC statistic was used for model selection. The models for species cover were also compared using ΔD, i.e. the difference between the deviances of the null model and the one selected. The species niche optima and probability of occurrence, tolerances and niche width of species along all coenoclines were estimated for the unimodal and skewed models (Lawesson & Oksanen, 2002).

3.3. Results

3.3.1 Relating species community composition to environmental variables

The final CCA model which included just the significant environmental variables (Table 3.2) had a total inertia of 2.288 of which 0.215 was accounted for by the significant environmental variables. The model was significant (pseudo-F =10.21; P< 0.001, %ΔAIC reduction from null model = 3.2%; from 3141 to 3039) and with eigenvalues of 0.105 and 0.074 for the first two axes. The environmental variables that were not included in the final model included aspect, cover of bare ground, litter, open water or animal excrement (used as a proxy for sheep grazing intensity) and vegetation total dry biomass.

The resultant species plot (Fig. 3.1a) shows the main dominant species *C.vulgaris* near the centre but offset from the main gradient. The main gradient (G1, Fig. 1a) reflects a transition from lichen-rich communities in the top left quadrant through to graminoid communities in the lower left quadrant. The offset of *C.vulgaris* from the main gradient is evidence of a secondary gradient (G2, Fig. 3.1a) reflecting elapsed time since burning, highlighting the importance of this species in the late-successional stages of the post-burn period.

The five moors show much overlap within the ordination space, although they all centre on *C.vulgaris* (Fig. 3.1b). Three of the moors (Broomhead, Midhope and Snailsden) had relatively small ellipses (9.4-11.4 units) and occupy more or less the same ordination space, but Howden and Bamford occupy much larger areas; 29.5 units and 38.0 units respectively. The species gradient from lichen- to grass-rich communities was highly correlated with spatial factors (easting and northing), elevation and soil pH, with pH reducing as elevation increases (Fig. 3.1c). There was also a second gradient from (top right – bottom left quadrants); this gradient reflects the impact of prescribed burning (burnt *C.vulgaris* either as stick or bush, both produced immediately after fire through to C.vulgaris; this gradient was highly correlated with elapsed time since burning and vegetation height (PD). Slope was highly correlated with the burnt *C.vulgaris* stick and bush variables, whereas moisture content was

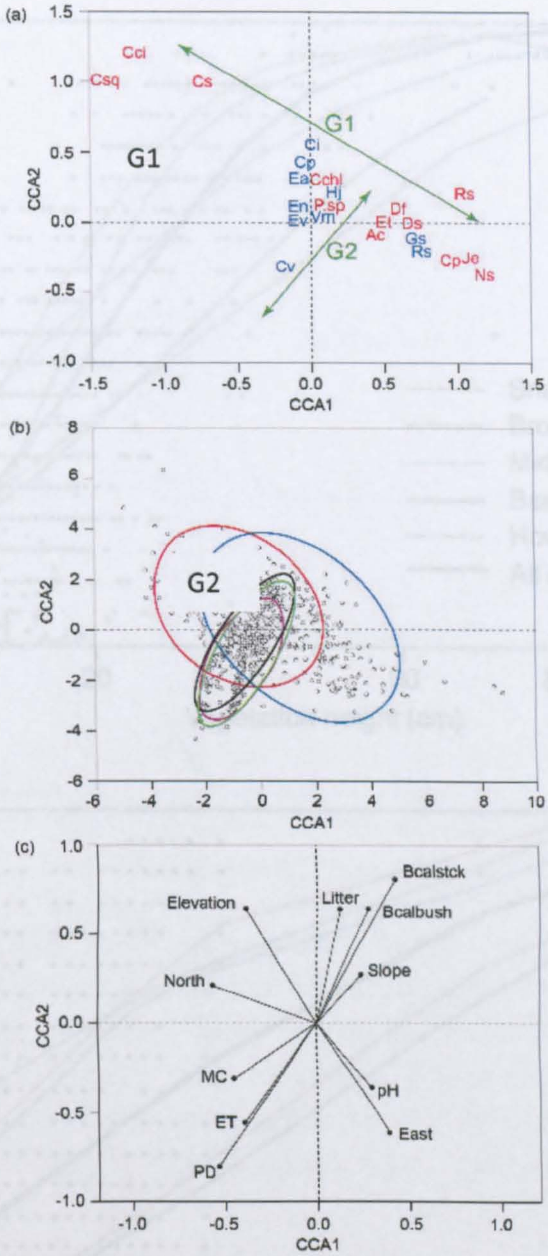

Fig. 3.1 Plots derived from the CCA analysis of plant species composition data and significant environmental variables. (a) Species plot, all species are illustrated, G1 and G2 highlight the major species gradients; (b) Quadrat plot identifying each Moor; using bivariate ellipses (SD with 95% confidence limits) superimposed; (c) Environmental variables. Codes are as follows: (a) Moors - Bamford =blue; Broomhead=green; Howden=red; Midhope=purple; Snailsden=black; (b) Species: most abundant species in blue; Ci=*Campylopus introflexus;* Cp=*Campylopus pyriformis*; Cv=*Calluna vulgaris*; Ea=*Eriophorum angustifolium*; Ev= *E.vaginatum*; En=*Empetrum nigrum*; Hj=*Hypnum jutlandicum*; Gs=*Galium saxatile*, Rc=*Rubus chamaemorus*, Vm=*Vaccinium myrtillus*; Less frequent species in red; Ac=*Agrostis capillaris*, Cci, Ccl, Cs, Csq= *Cladonia coccifera*, *C.chloropea*, *C.squamosa*, *C.*squammules respectively, Cpl-*Carex pilulifera*, Df=*Deschampsia flexuosa*, Ds=*Dicranum scoparium*, Et=*Erica tetralix*, Je=*Juncus effusus*, Ns=*Nardus stricta*, Ps=*Polytrichum commune*, Rs=*Rhytidiadelphus squarrosus*; (c) Significant environmental variables (Table 3. 2).

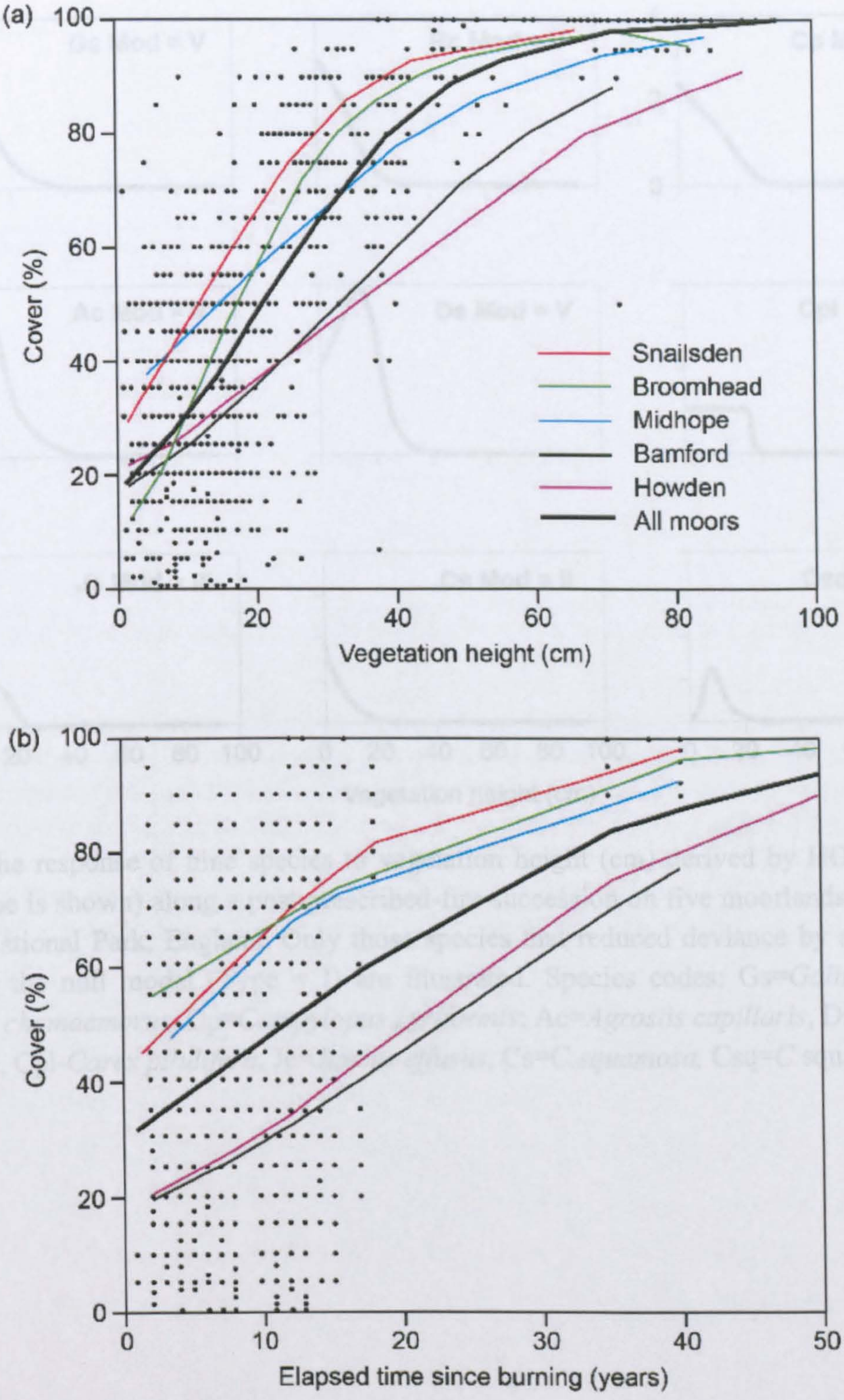

Fig. 3.2 The response of *Calluna vulgaris* to (a) vegetation height (cm) and (b) elapsed time since burning (years) derived by HOF-modeling along a post-prescribed-fire succession on five moorlands in the Peak District, National Park, England. The overall response is shown (bold line) along with the individual response curves for each moor. *C.vulgaris* exhibited a type II HOF model.

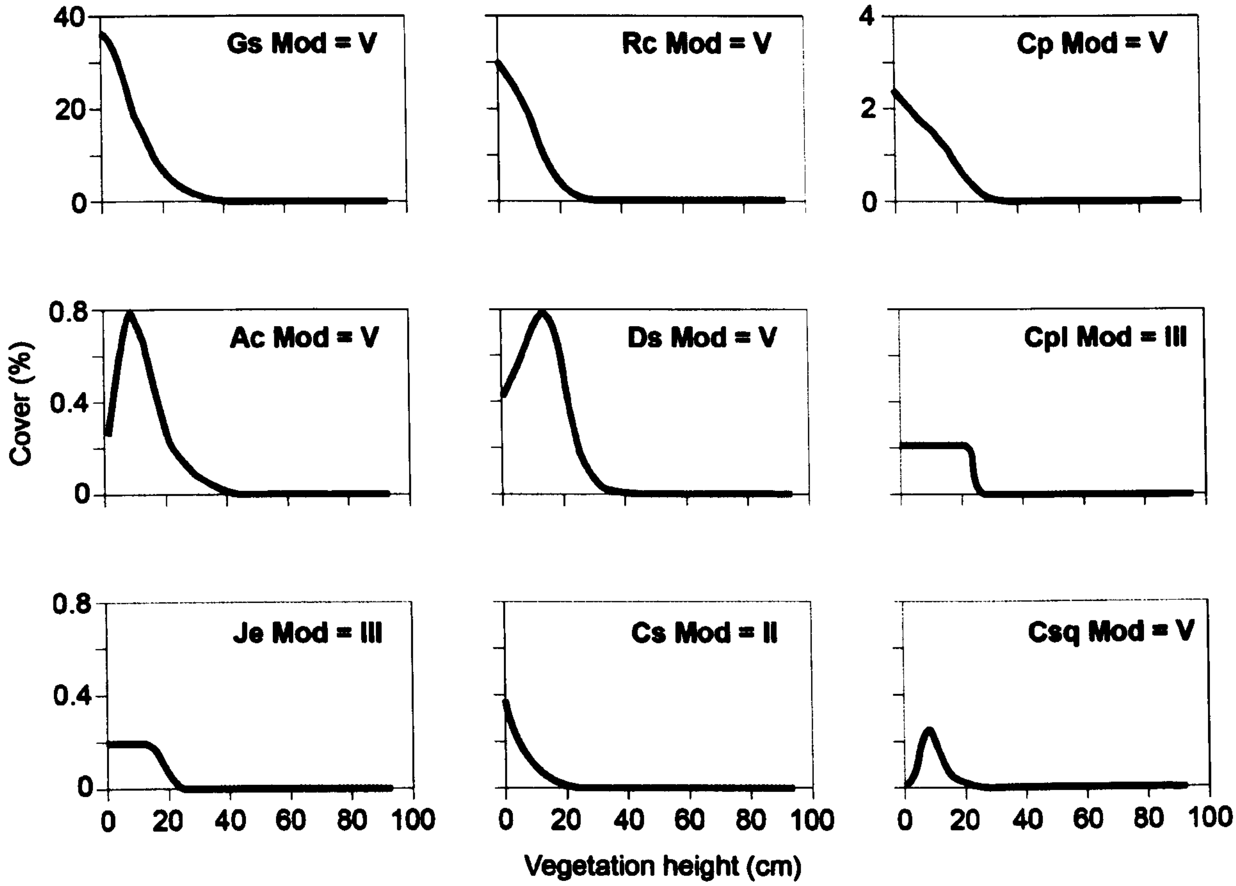

Fig. 3.3 The response of nine species to vegetation height (cm) derived by HOF-modeling (Model type is shown) along a post-prescribed-fire succession on five moorlands in the Peak District, National Park, England. Only those species that reduced deviance by at least 10% relative to the null model (Type = I) are illustrated. Species codes: Gs=*Galium saxatile*, Rc=*Rubus chamaemorus*, Cp=*Campylopus pyriformis*; Ac=*Agrostis capillaris*, Ds=*Dicranum scoparium*, Cpl-*Carex pilulifera*, Je=*Juncus effusus*, Cs=*C.squamosa*, Csq=*C.*squammules.

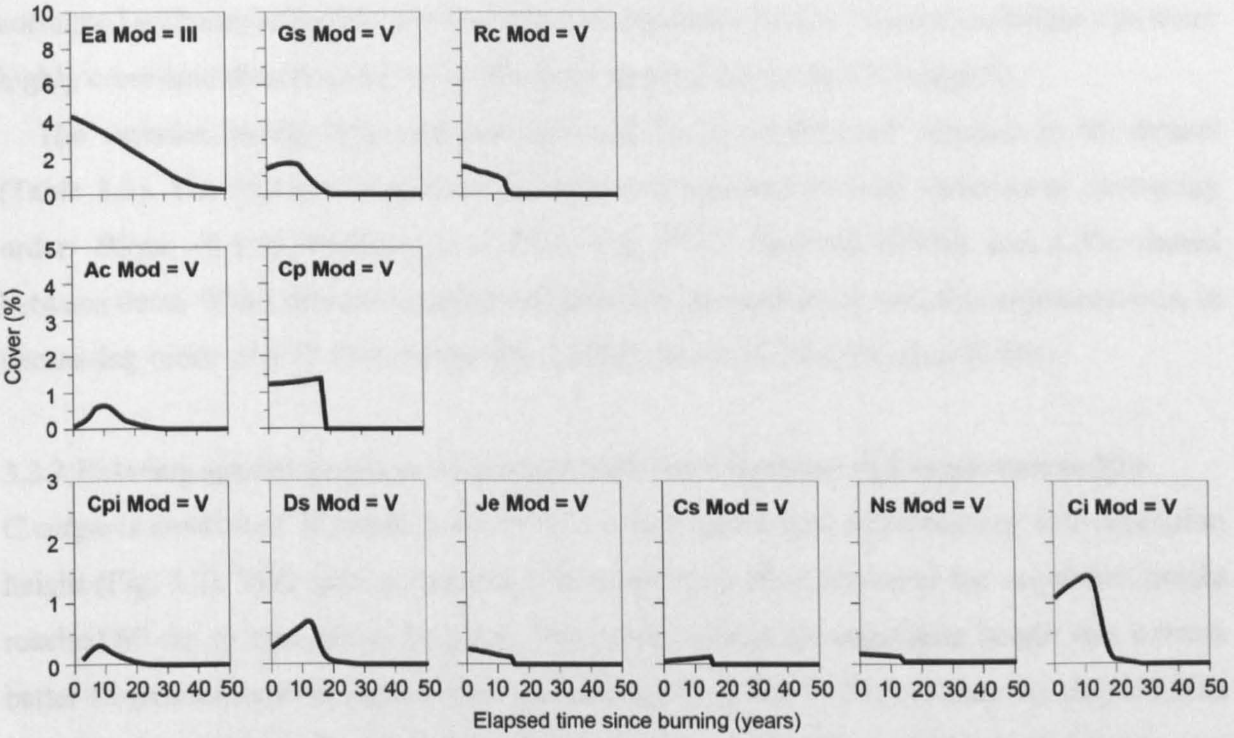

Fig. 3.4 The response of nine species to elapsed time since burning (years) derived by HOF-modeling (Model type is shown) along a post-prescribed-fire succession on five moorlands in the Peak District, National Park, England. Only those species that reduced deviance by at least 10% relative to the null model (Type = I) are illustrated. Species codes: Ea=*Eriophorum angustifolium*; Gs=*Galium saxatile*, Rc=*Rubus chamaemorus*, Ac=*Agrostis capillaris*, Cp=*Campylopus pyriformis*; Cpl=*Carex pilulifera*, Ds=*Dicranum scoparium*, Je=*Juncus effusus*, Cs=*C.squamosa,* Ns=*Nardus stricta*, Ci=*Campylopus introflexus.*

correlated with elapse timed since burning and vegetation height. Vegetation height was more highly correlated than elapsed time since burning with the cover of *C.vulgaris*.

The variation partitioning analysis explained 13.2% of the total variation in the dataset (Table 3.3). The four sets explained the following amounts of total variation in decreasing order: Biotic (8.1%), Production (7.2%), Site (5%), Physical (0.9%) and 6.5% shared between them. When shared variation was removed the amount of variation explained was, in decreasing order: Site (2.4%), Production (2.0%), Biotic (1.7%), Physical (0.6%).

3.3.2 Relating species response to elapsed time since burning and vegetation height

C.vulgaris showed an increase in cover with both elapsed time since burning and vegetation height (Fig. 3.2). This species reached > 90% cover on all moors after the vegetation height reached 60 cm or after about 30 years. The overall model for vegetation height was a much better fit (reduction in deviance from the null model, $\Delta Dev = 57.3\%$) than for elapsed time since burning ($\Delta Dev = 25.7\%$). Figure 3.2 shows the mean response and individual responses for *C.vulgaris* at each moor. The responses of Bamford and Howden are lower than the average and especially for the relationships with elapsed time.

All other species showed generally similar responses to vegetation height and elapsed time since burning, but only those species where the fitted response reduced the deviance by more than 10% are displayed (n=9 for vegetation height, n=11 for elapsed time since burning; Figs 33. & 3.4). All species either declined with respect to both variables, or peaked at the low end of the range. For vegetation height most species had declined to zero at between 20-40 cm in height and 20 years after burning, which was confirmed by inspection of the upper tolerance limit for those species that exhibited a Type IV and V response (Table 3.4). The upper tolerance values for almost all species were below 40 cm in height and 22 years since burning (the exceptions being *Empetrum nigrum* at 47 cm height, and *Eriophorum vaginatum* at 37 years after burning).

3.4. Discussion

There were several important results to come from this study, all of which provide information to help guide conservation policy with respect to the use of prescribed burning for moorland management. First, the results were derived from a multi-site study within a major region of the English uplands, and within each moor both patch and sampled quadrats were sampled strictly at random. Chronosequence studies such as these can of course be criticised because temporal change was inferred from stitching together a sequence of sites with differing histories. Our multi-site and sampling approach should help ensure that the overall picture produced from the results is consistent.

Second, the plant species community composition was relatively similar for the most part on all five moors sampled. This was somewhat surprising as the moors cover different elevations and the on-site management is applied slightly differently, as they are all managed through different ownership and management regimes. Nevertheless, in spite of an overall similarity in that all moors were centred around each other, there were major differences in the overall species pool on the five moors. This difference was evidenced by inspection of the ellipse areas; Bamford with the lowest elevational range had by far the greatest ellipse area, but here the ellipse showed that this moor encompassed the graminoid-dominated communities; Howden was intermediate and at its higher elevational range had a more lichen-rich vegetation, whereas the remaining three moorlands (Broomhead, Midhope, Snailsen) had the smallest species pools, and were all of similar size. The size of these ellipses reflected the species richness at each site. Moreover, these moors have a very limited species pool compared to potential target communities (British National Vegetation Classification; (Rodwell, 1991, 1992). This low diversity has been ascribed to past pollution especially since the start of the industrial revolution in the UK (Tallis, 1988; Ferguson & Lee, 1982), overgrazing (Anderson & Yalden, 1981), and intermittent, but very damaging wildfires (Albertson *et al.*, 2009). Why there should be such differences between moors may well reflect different past and current management practices.

The ordination analyses produced two gradients, the first relating to site factors including geography and elevation, presumably reflecting micro-climatic effects, and most of the variation was brought about by the two sites with the larger species pools. A much smaller gradient, which was centred on all sites, appeared to reflect the dominance of the major moorland species (*C. vulgaris*) and appeared to reflect the post-prescribed-burning succession.

Third, a major finding was that in the analysis of post-prescribed-burning species response or succession, *C.vulgaris* was the only species to increase in cover, producing more than 90% cover after the vegetation height reached 60 cm or after about 30 years. At this point *C.vulgaris* was the most dominant species, and this is reflected in the relatively large proportion of variation accounted for by the production variables, second only to site-level variables. All other species showed either a reduction or a unimodal response through the post-prescribed-burning succession time. For the most part these species had reduced to minimal levels by 22 years and 40 cm height with two exceptions, the exceptions being *Empetrum nigrum* at 47 cm height, and *Eriophorum vaginatum* at 37 years after burning. These results suggest that the disturbance caused by the prescribed burning creates gaps, with enhanced light levels and perhaps a flush of nutrients, and allows species to colonise from seed and/or resprout from stems, or rhizomes (Mallik & Gimingham, 1985). Thereafter, the *C.vulgaris* in particular, slowly becomes dominant and there is a decline in the cover of almost all other species. Species richness declined with elapsed time since burning and vegetation height on these five moors, and similar results have been found in single-site studies for plant species diversity (Hobbs & Gimingham, 1984) and lichens (Davies & Legg, 2008).

These results are consistent with the Intermediate-Disturbance hypothesis of Connell, 1978), where disturbance (here using prescribed fire) maintains greater plant species diversity than when left undisturbed. The evidence suggests that if burning is not carried out on a regular basis then the cover of all species detected here will decline. These moorlands are relatively species-poor as they have been subject to high atmospheric pollutant loads over the last two centuries, and from the species response curves presented here, there was no evidence detected to support the hypothesis that late-successional species will invade the moorlands if prescribed burning were not carried out. The results are also consistent with the initial floristic model (Egler, 1954), and both the tolerance and inhibition models of succession working together through time (Connell & Slatyer, 1977). The former model is supported by the increase in species cover of almost all species as the disturbance is created. The latter two models explain the observed changes in vegetation cover through all species (including *C.vulgaris*) expanding immediately after disturbance and thereafter all species other than *C.vulgaris* declining either as a result of their life-history strategies, or though competition from *C.vulgaris* as it shifts to being an inhibiting species. Nevertheless, prescribed burning will enhance and maintain the dominance of *C. vulgaris*, a feature that is not always considered a high conservation priority (McVean & Ratcliffe, 1962; Littlewood, *et al.*, 2010; Worrall, *et al.*, 2010). However, our results show that a no-burn practice on its own would not enhance the conservation status of these moorlands; rather it would make them less species-rich. Other intervention treatments, that will almost certainly involve species re-introductions, would be needed to complement a no-burn strategy.

3.5. Implications for practical reserve management

At least for the conservation of plant species diversity in the Peak District, the disturbance caused by prescribed burning appears to be critical, and the evidence suggests that if a no burning policy were implemented then there would be a reduction in biodiversity and an increase in the cover of mono-dominant stands of *C.vulgaris*. Recently, the emphasis on maintaining and enhancing ecosystem services implies that for carbon sequestration and water quality (many moorlands are water catchments) a reduction in burning frequency or indeed a no-burn policy would be better. Almost nothing is known about the impact of prescribed burning on carbon accounting (see Worrall, *et al.* 2009 for a discussion), but there is some evidence that burning may increase the amounts of Dissolved Organic Carbon (DOC) and certain nutrients (Yallop *et al.*, 2006; Yallop & Clutterbuck, 2009; Clay, *et al.*, 2010. This increase in coloration adds considerably to water utility purification costs.

However, the evidence here suggests that prescribed burning should be on a maximum of a 22 year rotation or at a maximum vegetation height of 40 cm. Given that field measurements of vegetation height, especially using the pressure disc approach used here, is easier then ageing *C.vulgaris* plants, it seems sensible to recommend that burning rotations should be applied somewhere between the 20-40 cm height, and using 40 cm as a maximum height guide.

Chapter 4

Impact of prescribed burning on moorland soil chemical properties in the Peak District, England

4. Impact of prescribed burning on moorland soil chemical properties in the Peak District, England

4.1. Introduction

The upland moors in Great Britain have a high conservation value of international significance (Ratcliffe & Thompson, 1988). They are dominated by dwarf shrubs, mainly *Calluna vulgaris* (L.) Hull, and are cultural landscapes being created and maintained by anthropogenic activity, which prevents succession to woodland and maintains a very nutrient-poor ecosystem (Gimingham, 1972; Webb, 1986; Marrs, 1993. These moorlands are managed mainly for sheep grazing and sporting interests (shooting red grouse, *Lagopus lagopus scoticus* (L.), and they contain six communities that are virtually confined to the UK (Thompson *et al.*, 1995). They also provide a habitat for other bird species such as curlew *Numenius arquata* (L.), golden plover *Pluvialis apricaria* (L.), black grouse *Tetrao tetrix* (L.), merlin (*Falco columbarius Linnaeus)*, hen harrier (*Circus cyaneus Linneaus)*, short-eared owl *Asio flammeus* (Pontoppidan) and ring ouzel *Turdus torquatus* (L.) (Robson, 1998; Whittingham, *et al.*, 2000, 2001; Thompson *et al.*, 1997). The perpetuation of these moorlands is, therefore, important in terms of both economy and biodiversity. Grouse shooting alone is worth 12% of the total UK shooting economy with and income of £120 million spent in the uplands during good years (PACEC, 2006).

Moorlands are managed in two main ways by sheep grazing and prescribed burning. Burning has been used for hundreds, perhaps thousands of years (Pearsall, 1950; Birks, 1988; Simmons, 2003), although it has increased in frequency over the last 150 years as a result of grouse management (Bonn, *et al.*, 2009). The prescribed burning is applied to patches on a rotation to produce a mosaic of stands in different stages of the burn-recovery cycle; the recovering *Calluna* produced forage for the red grouse and older, taller vegetation provides cover (Miller, 1980). The aim of the prescribed burning is to remove the woody plant growth and any old degenerate plants, leaving charred stems. The burnt stems resprouts to produce new growth, although regeneration reduces with plant age, and falls off sharply after ca. 15 years of age (Lovat, 1911; Miller & Miles, 1970). Where resprouting is slow, plant species need to regenerate from seed, which is either buried in the soil or disperses from outside the burn patch (Gimingham, 1994). Approximately, 65% of British upland moors are managed using this mosaic approach for the benefit of red grouse (Sotherton *et al.*, 2009).

Table 4.1 Details of the three study moorlands in the North Peak ESA, Derbyshire, England.

Moor	Grid reference (Longitude & Latitude)	Elevation range (m)	Range of patch ages since burning (yr)	Estimated age of reference patches since last burn (yr)
Bamford	SK 2287 8329 Latitude 53°21'N, Longitude1°40'W	300-420	4-16	40
Broomhead	SK 2449 9465 Latitude 53°27'N, Longitude1°38'W	300-460	3-17	52
Howden	SK 2013 9698 Latitude 53°28'N, Longitude1°42'W	272-540	4-17	52

Burning clearly will affect the nutrient content of the vegetation and will probably impinge on soil chemical properties. During burning, a large proportion of the nutrients within the standing vegetation may be released in more mobile forms, and may either be lost from the system in smoke, or deposited on the site as ash or char (Allen, 1964; Evans & Allen, 1972; Clay, 2009). Assessments of nutrient loss at different burning temperatures in the laboratory indicate that larger quantities of N are lost relative to other elements, although estimates are highly variable, ranging from 57% to almost a complete loss of N at temperatures above 500°C (Allen, 1964; Evans & Allen, 1972). Lower amounts of P and cations were volatilised with a fraction of these elements being retained in ash (White *et al.*, 1973). Deposition as ash provides a pulse of readily-available quantities of these nutrients which may, either be taken up by the developing vegetation, or lost through leaching or run-off (Clay, *et al.*, 2009 a, b). The amount being taken up by the vegetation will to some extent depend on the balance between rainfall patterns and the time taken for the vegetation to regenerate. As the vegetation recovers, there is an accumulation of biomass and litter (Chapman, 1967), and this will impact on soil chemistry through uptake and cycling of nutrients. Given the importance of prescribed burning and its potential impacts on nutrient release into waterways, it is surprising that almost nothing is known about its impact on soil chemical processes.

This paper, therefore, assesses the impact of prescribed burning of moorland on soil chemical properties within the Peak District Environmentally Sensitive Areas, Derbyshire in northern England (MAFF, 1993). Three hypotheses were tested: (1) Does burning have different effects on soils on different Moors, either as result of geographical location or management regime? (2) Is there a change in soil chemical properties with time after burning? (3) Is there an interaction with different temporal effects on the different moors? To test these hypotheses a chronosequence study was replicated on three separate moors. On each moor, patches which had been burned in different years (2-50/60 year sequence of elapsed time since burning) were selected randomly and soils sampled. Selected soil chemical properties were measured and both mixed-effects modelling and multivariate analysis were used to test these hypotheses.

Table 4.2 Soil characteristics of three moorlands in the North Peak ESA, Derbyshire, England compared with values derived from the literature for similar ecosystems. For the current data, arithmetic mean values ± standard errors (n=56) are presented along with the %ΔAIC statistic of the moor only mixed effects model relative to the null model (Appendices I,II). The model coefficients for Bamford were all significant ($P<0.001$), and where moors differed significantly from Bamford they are denoted as follows: *=$P<0.05$, **=$P<0.01$, ***=$P<0.001$. Literature ranges derived from (Allen, 1964) and (Allen *et al.*, 1969, 1989, Heal & Smith, 1978) for moorland soils and *(Marrs *et al.*, 1989) for infertile acidic upland grasslands; nd = not determined.

Soil property (units)	Moor: Bamford	Broomhead	Howden	%ΔAIC	Literature ranges
pH	4.12±0.02	4.08±0.02	3.94±0.02***	9.98	3.3-4.7
Bulk density (g cm^{-3})	1.09±0.02	1.10±0.02	1.01±0.01*	2.58	1.05-1.1
C:N (mol mol^{-1} on m^2 basis to 7cm depth)	29.2±0.44	33.9±0.9***	31.5±0.4*	2.0	
Total element concentrations					
NH_4-N (µg g^{-1})	25.5±2.5	3.3±0.8***	22.6±1.3	2.25	20-90*
NO_3-N (µg g^{-1})	3.3±0.8	0.5±0.2***	1.9±0.4	0.86	2-9*
P (µg g^{-1})	4.5±0.4	3.5±0.3**	4.9±0.2	0.94	2-21
K (µg g^{-1})	53.0±3.4	51.3±3.1	39.0±3.0*	0.14	25-780
Mg (µg g^{-1})	39.7±1.2	43.9±1.3*	39.5±1.0	0.18	11-500
Ca (µg g^{-1})	146.4±4.6	102.1±3.9***	86.2±3.2***	2.70	32-1300
Total element concentrations					
C (%)	35.1±1.2	23.1±1.7***	47.9±0.2***	3.86	6-50
N (%)	1.4±0.05	0.8±0.01***	1.8±0.02***	34.03	0.18-1.1
P (µg g^{-1})	642±32	381±25***	521±14	1.28	500-700
K (µg g^{-1})	317±11	342±24	271±11	0.57	nd
Mg (µg g^{-1})	274±11	203±13**	360±7***	1.46	nd
Ca (µg g^{-1})	1023±47	471±37***	1085±25	1.79	nd

Table 4.3 Predicted nitrogen balance in three moorlands in the North Peak ESA, Derbyshire, England based on measured changes in C:N ratio over a 50 year period., Predictions are made using equations derived from literature sources (Min = lower estimate from Gundersen *et al.*, 1998; Max= maximum estimate from Jenkins *et al.* 2001). Equations: Min= 2.0371-0.0638*C:N (r^2=0.71); Max= 1.4995 -0.0377*C:N (modelled line – hence no r^2). Note positive = nitrate loss, 0 = all nitrogen inputs captured.

Moor	C:N ratio Constant Year=0	N Out/In flux		C:N ratio predicted Year=50	N Out/In flux	
		Min	Max		Min	Max
Bamford	30.2	0.11	0.36	27.5	0.28	0.46
Broomhead	35.1	0	0.18	32.4	0	0.28
Howden	32.7	0	0.27	30.0	0.12	0.37

4.2. Methods

In order to provide an estimate of the range of variation within the region this study was performed on three replicate moors within the North Peak ESA, Derbyshire. The three moors (Bamford, Broomhead and Howden, Table 1) vary in elevation (272-540 m), ownership and management styles. Within each moor, all burn patches were digitised from aerial photography and linked to information on age of burn determined from the records kept for management subsidy audit. Ten patches were sampled randomly from each moor to provide a chronosequence of patches which had been burned at varying times in the last 20 years plus three patches which had not been burned for at least 40 years. Each patch was then overlain with a geo-referenced 1m^2 grid within (ArcMap, 2008), and four 1 m^2 sampling positions were then selected randomly (total number of sampling positions=156).

In April 2009, each of the sampling positions was located using two (Garmin eTrex, GPS) units, which were calibrated at National Grid Trigonometry points. Correction factors were derived and averaged, and the applied in the field to ensure accurate location of sampling positions. At each position, the cover of each species of higher plant and bryophyte plus the presence of *Cladonia* lichens was recorded in a 1m^2 quadrat. The vegetation from the central 25 x 25 cm area was cut and separated into living material and litter. Five soil cores (5cm diameter, 7cm depth) were taken from each corner of the 1m^2 quadrat and the centre, pooled and stored at ~5°C. A sub-sample (ca. 100g) of each soil sample was removed for chemical analysis; half was air-dried to a constant mass and then finely ground, the remainder was stored as a fresh sample at ~5°C. Bulk density was estimated by dividing the mass of the entire sample by the known total volume of the 5 soil cores.

4.2.1. Chemical analysis

Soil pH and plant-available concentrations of selected nutrient elements were determined on the fresh soil. Soil pH was measured in a 1:2.5 suspension of soil and deionised water. For extractable nitrogen fractions 5 g of soil was extracted in 50ml 2M KCl and shaken for 60 minutes, centrifuged for 5 minutes at 4000 rpm and filtered. For plant-available P, K, Na, Ca and Mg concentrations, a similar extraction procedure was used, but with 2.5% acetic acid as the extractant (Allen *et al.*, 1989). Total element concentrations were determined on the air-dried soil sub-sample. C and N concentrations were measured directly on the soil samples using a Carlo Erba Instruments NC2500 elemental analyser. For P and the cations, sub-samples (0.5g) were ashed in a muffle furnace at 550°C for 2.5 hours and the ash taken up in 10 ml of 2M HCl (Allen, *et al.*, 1989) and made up to 50 ml.

The concentrations of ammonium-nitrogen, nitrate-nitrogen and P concentrations were determined using a Bran & Luebbe AutoAnalyzer 3 (Bran & Luebbe, 2006). Potassium and were estimated by emission spectrophotometry and Ca and Mg by absorption spectrophotometry (Allen *et al.*, 1989). Elemental concentrations were expressed on a concentration basis (μg g^{-1}) and a volumetric basis (μg cm^{-3}). The C:N ratio was analyzed on both a g g^{-1}, and a mol mol^{-1} basis calculated on a m^{-2} basis to 7 cm depth. The latter measure is used here to be comparable with published data elsewhere.

4.2.2. Data Analysis

First, the results for each soil variable were analysed using mixed-effects models (Crawley, 2007). Both moor and elapsed time since burning were included as fixed factors in the model, and patches within each moor were treated as random effects. This approach accommodated the hierarchical nature of the study design, and allowed five models to be tested using the model deletion approach to derive the Minimum Adequate Model (MAM), (Crawley, 2007); the models were (i) the interaction model (moor x elapsed time), (2) the additive model (moor + elapsed time), the moor only model, the elapsed time only model, and the null model. Terms were deleted sequentially using the Maximum Likelihood method, and thereafter, the selected MAM was re-run using the Restricted Maximum Likelihood Method (REML) (Crawley, 2007). These analyses were implemented in the R software environment (R Development Core Team, 2009), using the NLME package for Linear Mixed Models (Pinheiro & Bates, 1996; Venables & Ripley, 2002). As the outputs from the analyses of the concentration and volumetric data (Rosenburgh, 2009) were similar, only the concentration data are discussed in detail here. A completely annotated analysis along with all statistical output is available in (Rosenburgh, 2009).

Second, the combined data for all soil variables (pH, bulk density, available concentrations, total concentrations, which were corrected by subtracting the available concentrations) were analysed by both Principal Components Analysis (PCA) and Redundancy Analysis (RDA). In both analyses the soil variables were standardised to zero mean and unit standard deviation. In the RDA the soil x quadrat matrix was correlated with environmental variables (moor, vegetation height, elevation, moisture content and elapsed time since burning). Within the RDA the forward selection was used to select the most significant environmental variables using the AIC statistic as the criterion. Finally, the moor positions on the resultant biplots were displayed as bivariate-standard-deviational ellipses. All of these analyses were performed using the 'rda' and 'ordiellipse' functions in the vegan package (Oksanen, 2005) in the R software environment (R Development Core Team, 2009). The outputs form both PCA and RDA were similar; only the RDA is reported here.

4.3. Results

4.3.1. Analysis of individual soil variables

In these analyses only three models were selected as significant: (1) the moor only model, where only the constants were significantly different, indicating differences between moors; (2) the additive model (moor + elapsed time since burning), where the constants were significantly different, but the slopes were the same, here the moors were different from each other but the response to time was common, and (3) the interaction model (moor x elapsed time since burning), where both constants and slopes were significantly different, i.e. the moors were inherently different (constants) and the response to time was also different at each moor. The results of the statistical analyses are presented in.

Essentially, these analyses showed that all measured variables differed between moors, but for some variables there was either a common temporal effect between moors, or there was a different temporal effect on each moor with respect to time since burning.

4.3.2. Differences in soil properties between moors (Table 2)

The soils were typical acidic organic moorland soils elsewhere in GB. Almost all variables were in similar ranges to these reported for uplands soils, albeit at the lower end of published ranges for the available nutrients (N, P, Ca, Mg). The exception was total N which had values at the mid-upper end of the published range, especially for Howden.

Bulk density and soil pH differed between the moors, with Bamford and Broomhead having similar values (1.1 g cm^{-3}; pH=4.1); and Howden being slightly lower (1.0 g cm^{-3}; pH=3.9). Ammonium-N was the most abundant form of available N, at least seven times greater than the amount detected as nitrate-N, For both forms of available N and available P, Bamford and Howden had similar values, 22.5-25.5 μg NH_4^+-N g^{-1}, 2-2.3 μg NO_3^--N g^{-1} and 4.5-4.9 μg P g^{-1} respectively, but Broomhead had significantly lower values (3.3 μg NH_4^+-N g^{-1}; 0.5 μg NO_3^--N g^{-1}; 3.5 μg P g^{-1}), For Mg, the same groupings of moors was found, but here Broomhead had greater values 44.5 μg Mg g^{-1} than the other two moors (39 μg Mg g^{-1}). K concentrations followed the pH results with Howden having lower values 39 µg K g^{-1} than the other two moors (51-53 µg K g^{-1}). Ca showed the greatest differences between moors,

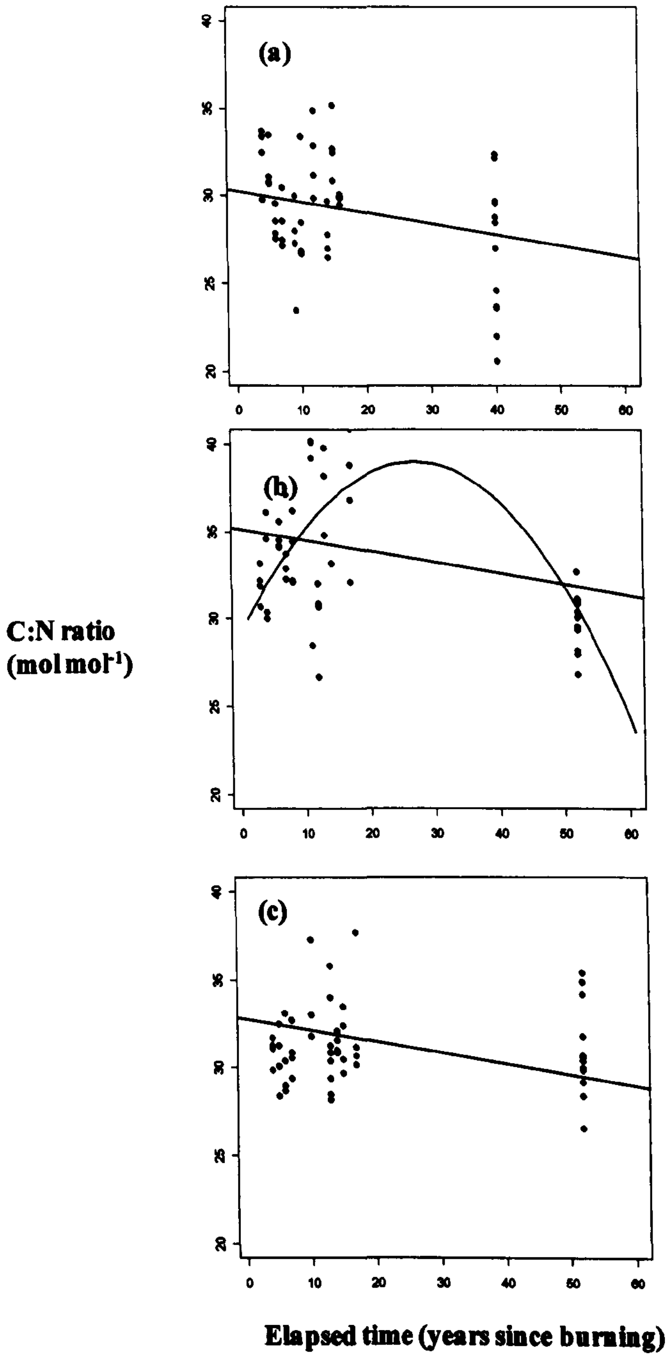

Fig. 4.1 The C:N ratio in soils (mol mol^{-1} expressed on g m^{-2} per 7 cm depth) along a chronosequence on three moors in the Peak District, Derbyshire. Mixed-effects models identified significant differences between moors, but there was also a significant common temporal response at each moor. (a) Bamford; (b) Broomhead; (c) Howden. Broomhead was the only moor that exhibited a significant non-linear response; Linear fit - $r2=0.14$, $F1,50 = 9.52$, $P<0.003$; quadratic fit – $r2=0.36$, $F2,49=15.3$, $P<0.0001$.

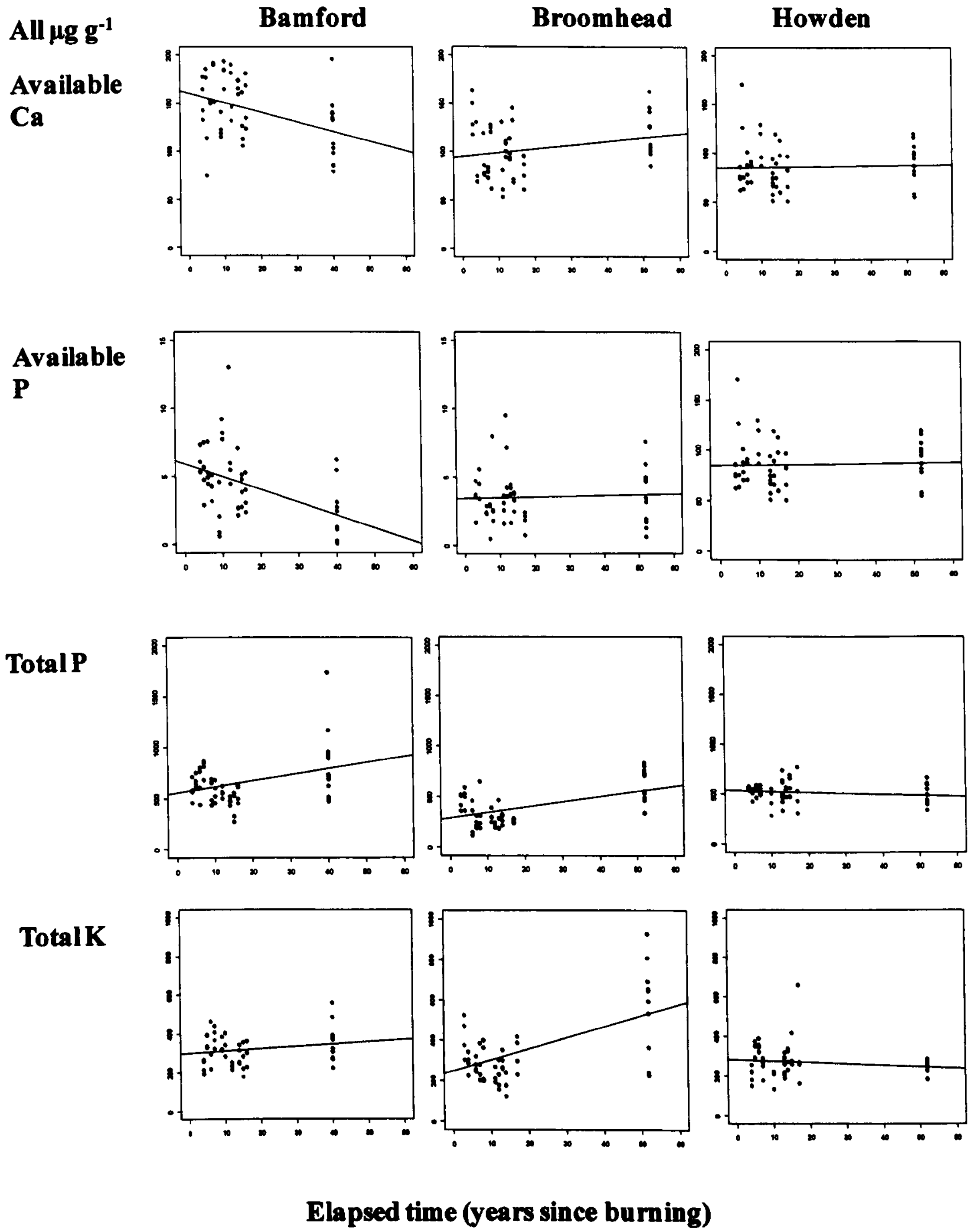

Fig. 4.2 The concentrations of available and total elements that showed significant differences between moors in their interaction with elapsed time since burning (significant constants and slopes).

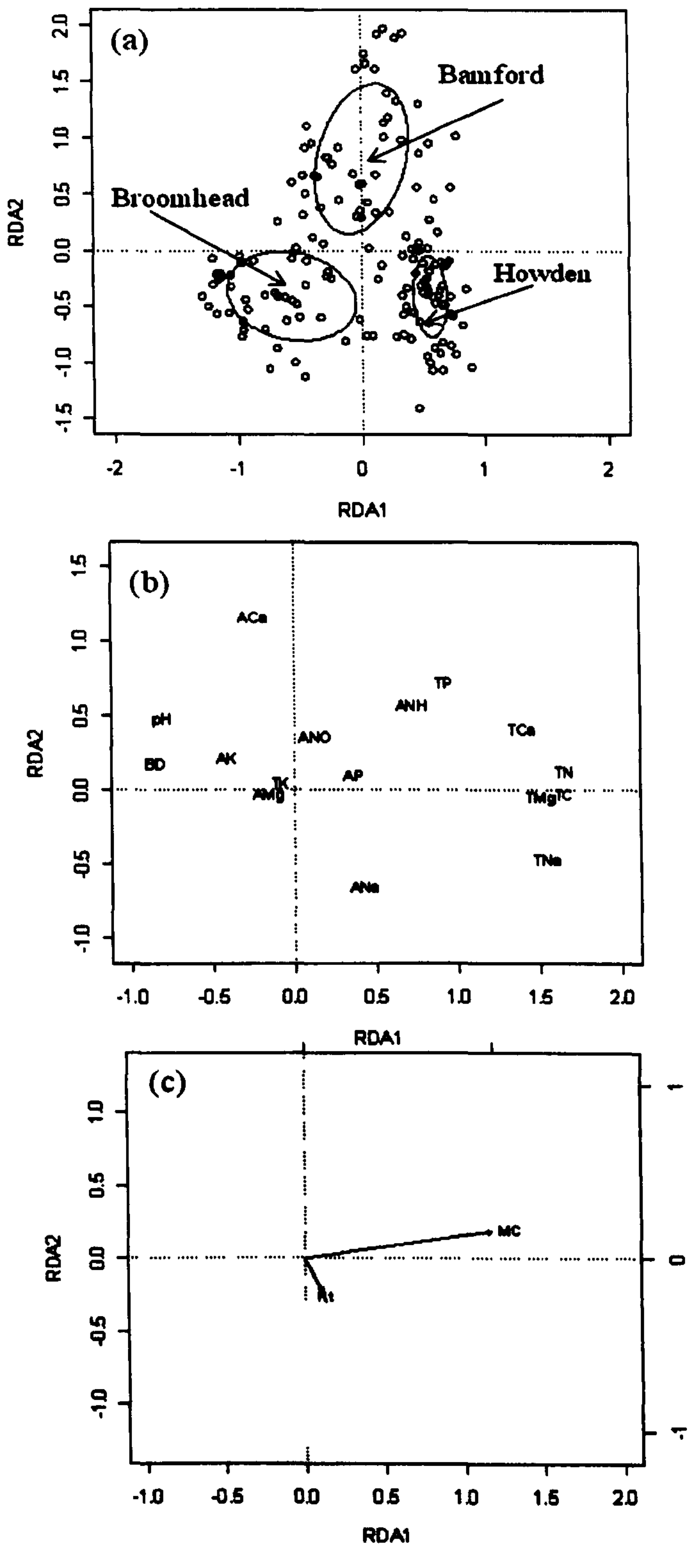

Fig. 4.3 The RDA biplots of soil concentrations along chronosequences after burning on three moors in the Peak District, England. (a) individual quadrats and the distribution of the three moors expressed as bivariate-standard-deviational ellipses, (b) soil variables, and (c) the three significant environmental variables (MC = Moisture Content, Ht = vegetation height). All graphs are from the same analysis. Key to soil variables: BD = Bulk density, prefix A = available, prefix T = Total, elements given as usual abbreviations.

Bamford had the greatest concentration, Broomhead was intermediate and Howden had the least, 146, 102 and μg Ca g^{-1} respectively.

Soil C and N both showed a gradient with Howden having the greatest total C and N concentrations (C = 47.9%, N = 1.79%), Bamford being intermediate (C = 35.1%, N = 1.41%) and Broomhead the least(C = 23.1%, N = 0.80%). The C:N ratios, however, did not follow this pattern as the lowest ratio was found at Bamford (26%), Howden (28%) was intermediate Broomhead had the greatest (30%). The total amounts of the other elements showed that Broomhead had significantly lower concentrations of P and Ca than the other two moors (381 μg P g^{-1} and 471 μg Ca g^{-1} compared with 521-641 μg P g^{-1} and 1023-1085 μg Ca g^{-1}) , and for Mg there was a gradation from Howden (360 μg Mg g^{-1}) with the greatest values, Bamford being intermediate (274 μg Mg g^{-1}) & Broomhead the least (203μg Mg g^{-1}). There was no significant difference in total K concentrations between moors.

4.3.3. Soil variables where there was an additional common temporal response after burning

The ratio between carbon and nitrogen showed a significant moor + elapsed time effect (Fig. 4.1). Each moor had a significantly different intercept between all moors (Bamford = 30.2, Broomhead = 35.1, Howden = 32.7), which compared favourably with the arithmetic moor averages (Bamford = 29.2, Broomhead = 33.9, Howden = 31.5), but with the same rate of decline (-0.06232 yr^{-1}) over the course of the chronosequence on all moors. Linear and quadratic equations were fitted to each of the moors individually and Broomhead was the only moor to show a non-linear response (Fig. 4.1).

4.3.4. Soil variables where there were additional different change after burning on each moor

Bamford and Broomhead had similar intercept values for available P, which were both greater than Howden, but the moors had very different slopes; Bamford declining sharply with age, Broomhead showing a slight decline and Howden showing little change (Fig. 4.2a). For available Ca, Bamford had a much greater intercept than the other moors but this declined sharply with elapsed time since burning; Broomhead and Howden had similar intercepts, and while Howden showed no change with elapsed time, Broomhead increased its concentration through time (Fig. 4.2b).

The total concentrations of P and K showed different responses. For P, Bamford and Howden had similar P intercepts, with Broomhead being significantly lower, (Fig.4.2c), whereas for K all three moors had similar intercepts (Fig. 4.2d). The response to elapsed time since burning showed similar responses for these two elements. P and K increased with time on Bamford and Broomhead but reduced slightly on Howden. For P Bamford and Broomhead showed almost parallel increases with time, for K the increases at Bamford were much lower than at Broomhead. Interestingly, for P opposite signs of temporal responses were found for available and total P concentrations on the different Moors. Bamford and Howden showed a decrease in available and an increase in total P with time since burning and Broomhead showed the opposite effect.

4.3.5. Multivariate analysis of soil chemical properties

The null RDA model had an AIC value of 433.52 and three variables were selected as significant, reducing the AIC 357.68 (17.5% reduction). The three significant variables were moor, moisture content and vegetation height; notable elevation and elapsed time since burning were not significant. The constrained RDA variables accounted for 41% of the variation, and the distribution in relation to moors (Fig. 4.3a) shows a clear separation of the three moors in the ordination space. This is linked to the soil variables (Fig. 4.3b), with soil pH and bulk density at the negative end of axis 1, along with the available nutrients around the origin and the total concentrations at the positive end. On axis 2 the gradient reflects higher pH, available calcium at the positive end and both available and total Na at the negative end. The links to the significant environmental variables show that moisture content is correlated with axis 1 and vegetation height with axis 2, albeit weakly. Interestingly the high pH and available nutrients are in the area of the space occupied by Bamford and Broomhead, Howden is located where the vegetation height is greater and is correlated. The moister soils, have high Na and high total concentrations of nutrients and vegetation height (Fig, 4.3c). The general separation of the moors is probably based on altitudinal range (Howden reaches the highest elevations) where the soils are moister and a greater Na concentration might be expected through higher inputs from rainfall.

4.4. Discussion

As moorland vegetation only exists on very infertile soils within a cultural, managed landscape (Gimingham, 1972, 1996) any management that affects soils chemical properties needs to be investigated. Clearly, prescribed burning, which transfers nutrients from the stored pool in the vegetation could impinge on long-term ecosystem persistence (Marrs, 1993). Three important results were detected in this chronosequence study of the effects of prescribed burning on moorland soils. First, for the majority of soil chemical properties examined the only significant difference was between the replicated moorlands. Second, the impact of prescribed burning and subsequent vegetation recovery had little effect on most of the soil chemical properties measured. Third, some chemical properties did show a temporal relationship with the burn-recovery chronosequence cycle, and two types of response were found, an additive response for the soils C:N ratio between moorlands, and an interaction with the three moors for four soils properties (available Ca, P, total P, K). The most interesting, and indeed worrying result from an ecological viewpoint was the impact on C:N ratio through time.

4.4.1. Differences between moors

The soil chemical properties of all three moors fell, for the most part, within published ranges for upland moorland and grassland soils in Great Britain. However, within these typical ranges significant differences in all soil properties were detected between the three moors by both the univariate and multivariate statistical methods. Some variables, albeit significantly different, showed almost trivial differences (soil pH and bulk density) but the others showed considerable differences between moors. Broomhead, for example, was particularly low in total soil C, N and available N and P, whereas Howden had high total Ca and Mg but low exchangeable concentrations. Interestingly the environmental factors selected as significant in the multivariate analyses were vegetation height and soil moisture content, almost certainly related to site elevation and rainfall. That these peaty soils should vary so much when they are so close geographically (within 5 km of each other) develop on similar substratum (millstone grit), have been formed by the same suite of species (mainly Calluna vulgaris), have a low grazing pressure is in itself interesting. The sites are managed differently, but they are all subject to low-level sheep grazing (ESA prescription grazing pressure in force since 1994, is currently 0.5 sheep ha^{-1} summer grazing (Pakeman *et al.*, 2000), and prescribed burning. What is important from a conservation management viewpoint is that prescribed burning had no significant effect on most soil chemical properties, the exceptions are discussed below.

4.4.2. Different responses through time on different moors (interactions)

Given the relative similarity of the moorlands, the interactions responses were surprising. Howden showed least effect for all of the four soil significant interactions (available P, Ca, total P, K) with an almost flat response, but the other two moors showed different and sometimes opposite relationships with elapsed time since burning. The available P and Ca at Bamford were much greater immediately after burning, perhaps indicating that either greater quantities of these elements on this site are deposited as ash, or that they are not subject to high leaching immediately after the fire but decline slowly either through leaching or plant uptake (Allen, 1964; Allen *et al.*, 1969). At Broomhead this hypothesis might apply for P but not available Ca which increased during the chronosequence. This aggrading responses was also seen at Bamford and Broomhead for total P and Ca and these increases can presumably only be derived from a combination of aerial deposition from rainfall, perhaps augmented by ash deposition from adjacent burns, or there is increased cycling so that these elements are preferentially taken up from lower layers and released into the surface layer. As *Calluna* is the dominant species on these moors and is shallow-rooted this latter hypothesis seems unlikely (Gimingham, 1960).

4.4.3. The reduction in C:N ratio with time on the different moors

The C:N ratio was significantly different on the three moors (projected constants: 30.2, 35.1, 32.7 for Bamford, Broomhead and Howden respectively) but declined at the same rate on all three moors (-0.06232 yr^{-1}). This result is counter-intuitive because it would be expected that in the older stands, biomass and litter would accumulate (Chapman, 1967). In these ecosystems the surface soil is highly organic mor humus and is classified as a peat (cf. descriptions of similar soils by (Heal & Smith, 1978). Hence, the expectation would be that decomposition would be slow and that organic matter would transfer from the litter into the soil. The C:N ratio is a key variable influencing decomposition (Swift *et al.*, 1979; Gundersen *et al.*, 1998), where the C:N ratio is low (<12:1) then decomposition is rapid. As the C:N ratio increases decomposition declines and peat will be increasingly formed. In upland areas of Great Britain, this ratio has been suggested as a reasonably proxy for decomposition processes leading to nitrate leaching expressed as the ratio of N out/ N in from catchment studies (Gundersen *et al.*, 1998; Jenkins *et al.*, 2001). The critical range for no nitrate leaching in these studies appears is between 32:1 and 40:1 (Table 3). In the three moorlands studied here, Broomhead and Howden are within this range and Bamford is just below it, but the rank order in terms of risk was Broomhead < Howden < Bamford. However, all of the moors showed a decline over time, the very oldest stands implying that after a period of no burning the C:N ratio declines with risk from decomposition and nitrate losses. Using regression equations derived from (Gundersen *et al.*, 1998; Jenkins *et al.*, 2001) predicted changes in N balance were calculated for Year = 0 and a 50-year no burn period (Table 3). These estimates indicate that at the minimum estimates Year = 0 range from zero at Broomhead and Howden to a small flux (0.11) at Bamford to between 0.18-0.36. After 50 years with no burning the minimum estimates remain zero at Howden but both other moors would be releasing nitrogen (flux=0.12-0.28), and the maximum estimates indicate all sites would leach N (flux=0.28-0.46). The discrepancies between the two studies reflect differences in catchments, locations and methods of calculations. There is also the inherent uncertainty in up scaling from the plot scale studies here to catchment scale models. Nevertheless, these projected losses are presented as indicative of the changing processes that are likely to occur if these moorlands are not burned.

4.4.4. Weaknesses in this approach

The obvious weakness in this research is the length of time span available for developing the chronosequence, especially in the older stands. The reason for this is that most of the moorlands are burned on about a 10-20 year rotation so most patches are in this age period. Older stands result from previous large scale burns, sometime wildfires, and these areas have been allowed to recover without any subsequent burning (G. Eyre, pers. comm.,). These remnants are small and scattered. This approach, therefore, can be criticised for the lack of sites in the 20-40 year period, a crucial period in the dataset. In addition, it could be argued that the oldest stands must be atypical because they have not been burned in recent times. This is possible, and that is why three replicates were taken to assess any potential variability. Moreover, we have restricted our analysis to mixed-effects linear models, and at least for some of the individual relationship it might have been appropriate to fit non-linear models. The non-linear relationship between C:N ratio at Broomhead lends weight to this argument, but our results also indicate that a linear relationship was valid for this measure at the two other sites.. Irrespective, the older stands still have a much lower C:N ratio, although clearly more research is needed to clarify these relationships.

Furthermore, the oldest stands would have been growing vigorously during the period when atmospheric nitrogen pollution was at its greatest; in this geographic area atmospheric deposition on NO_2 has declined by ca. 30-50% since 1985 (Anon, 2009), and it is possible that elevated N added to the system decreased the C:N ratio of the plant tissues in these stands and this has been transferred to the soils. *Calluna* and three other species typical of wet, upland ecosystems, and found in a three moor syudy (*Deschampsia flexuosa* (L.) Trin., *Hylocomium splendens (*Hedw) Br. Eur., *Nardus stricta* L.) show a positive relationship between atmospheric N depositions and foliar N concentration (Hicks *et al.*, 2000). In the younger stands any N accumulation would have been mitigated through the burning process. These potential issues need to be addressed by further work and modelling.

Chapter 5

Factors affecting biomass reduction in prescribed fires on upland moorland

5. Factors affecting biomass reduction in prescribed fires on upland moorland

5.1. Introduction

Prescribed burning is a traditional management practice carried out on UK moorlands to maintain productivity of the dominant species, heather, *Calluna vulgaris* (L.) Hull, for sheep grazing, red grouse (*Lagopus lagopus scoticus* (L.) production (Gimingham, 1972) and wildfire prevention (Worrall, *et al.*, 2010). As many of these moorlands have a high conservation value (Thompson, *et al.*, 1995, Ratcliffe & Thompson, 1988), and provide considerable ecosystem services (carbon storage, water catchments, recreational activity; (Tucker, 2003), there is a need to develop good practice guidelines for prescribed moorland burning. Prescribed burning is controlled by legislation and codes of good practice which differ regionally within Great Britain; (Defra, 1997; SEERAD, 2001a, b); Welsh Assembly Government, 2008) but which all restrict prescribed burning to defined periods in the year. In the English uplands, for example the permitted burning period is from 1st October to the 15th April (Defra, 2007).

Where the aim is to maximise *Calluna* regeneration, prescribed burning should ideally remove all above. Ground vegetation, including moss (Davies, *et al.*, 2010), whilst minimising damage to roots and soil. This is equivalent to "light" fire severity sensu (Keeley, 2009; adapted from Ryan & Noste, 1985), whereby surface litter, mosses and herbs are charred or consumed but the soil organic layer remains largely intact. Prescribed burns in the UK are traditionally lit using a naked flame from a wick or "fire kettle". However, this approach only works well with relatively dry vegetation and is thus limited by weather. The number of days that burning can be carried out is, therefore, limited, and even when burning is possible, burns can only be started late in the day. More recently, an approach has been developed where pressurised diesel or gas is sprayed on to the ignition site before lighting. This technique has been referred to as "cool" burning" but it is perhaps better described as "pressurised-fuel-assisted", or "PFA" burning. This technique was developed to achieve biomass removal without damage to underlying peat and leaving the bryophyte layer charred, but otherwise intact. The PFA burning technique allows prescribed fires to be lit during wetter conditions, and hence increases the number of days within the burning season when burning can be implemented. Regeneration occurs from resprouting *Calluna* within the remaining bryophyte layer within the first summer. Where the bryophyte layer is wholly removed regeneration tends to be slower (G Eyre, unpublished)

This study considers the relationship between fire severity (assessed through the amount of biomass removed) and (a) fire characteristics, and (b) environmental variables in prescribed burns.

5.2. Methods

A total of 17 prescribed fires implemented by the land manager as part of the normal burning regime on Howden Moor, Peak District, UK was studied National grid reference; Longitude1°41'W, Latitude 53°41'N, (BNGR) between March and April in 2007, 2008 and 2009. Within each area to be burned, four measurement points were positioned in a 'zig-zag' pattern (Fig. 5.1). Before burning, the cover of plant species was estimated within a 50 x 50 cm quadrat at each measurement point. The vegetation was then removed to ground level using secateurs and sealed in airtight bags until weighed. A second quadrat, adjacent to the first was located in a random direction from the first and a thermocouple Omega thermocouple K, Range -200°C to 1,200 °C; accuracy ±0.5 °C, OMEGA Engineering Ltd, Manchester, UK attached to the base of vegetation in the centre (ca. 1cm from ground with sensor not touching vegetation). A second thermocouple was located directly above the first thermocouple, just below the vegetation canopy. There were, therefore, 8 thermocouples in each burn (4 positions x 2 heights). The maximum height of vegetation was recorded in the centre of every quadrat along with slope and aspect.

The plot was then burned by the land manager; a line of vegetation approximately 16-20m wide and at least 2 m upwind from the first sensor (Fig. 5.1) was sprayed with diesel and then lit with a flaming torch. The usual measures to control the fire were used; these included the use of fire beaters and fire foggers (http://www.firefighting.co.uk). The temperature was recorded at each thermocouple every second by an eight-channel data logger Omega OM-DAQPRO-5300, OMEGA Engineering Ltd, Manchester, UK that was positioned at least 1.5 m from the edge of the burn. Once cooled, vegetation from the secondary quadrat was removed and bagged as described above. Pre- and post-burn vegetation were separated into fractions (*Calluna,* litter, moss, and animal excrement) and weighed. Mass was recorded before and after drying in an oven until a constant mass was reached, thus giving fresh and dry mass values for each fraction. Temperature measurements were discarded from some locations because either a thermocouple malfunctioned or part of the intended burn area escaped the fire. The final dataset contained 53 locations across the 17 prescribed burns. All data from the field was collected underwritten quality control protocols.

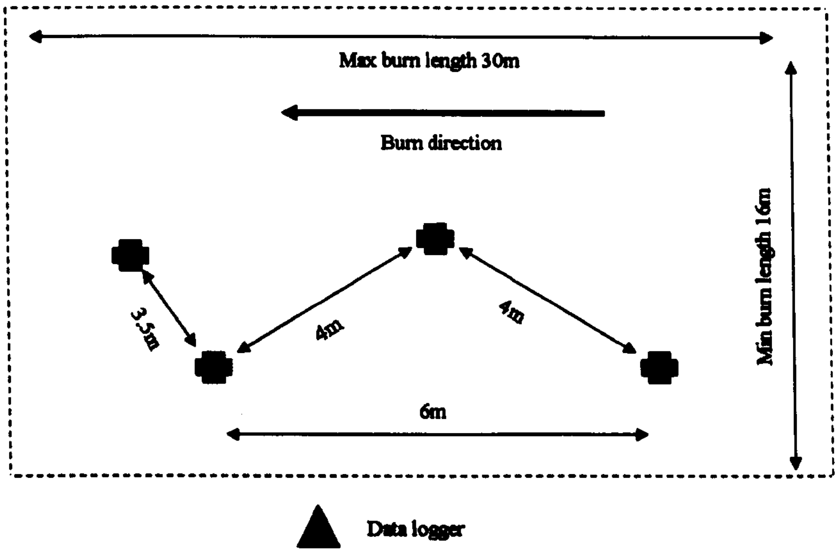

Fig. 5.1. Positioning of quadrats and thermocouples (crosses) within the burned area (dotted line) and data logger location.

5.2.1. Data analysis

Maximum temperature (°C) and the burn start and end points were extracted from each burn profile (Fig. 5.2). Burn start and end points were defined as the first and last time the temperature exceeded 60°C; this temperature was chosen as a conservative estimate of the point of plant cell thermal death (Yan & Hunt, 1999; Wahid *et al.*, 2007). Burn residence time or duration (s) was calculated as the time (s) between start and end. Total heat (°s) experienced was calculated as the area under the temperature curve between these two points. Each of these characteristics was recorded for each thermocouple location.

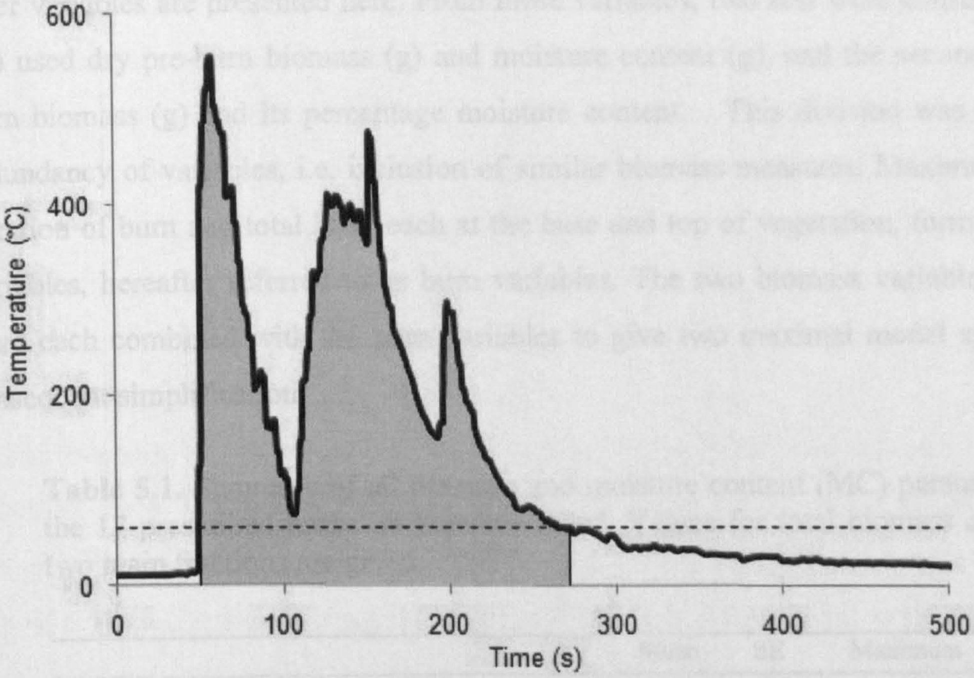

Fig. 5.2. An example representative burn profile from thermocouples within a prescribed burn (27th March 2007; solid black line). Also shown are the start (a) and end (b) points of the burn (first and last times where temperature exceeded 60°C, dotted line); burn residence time (number of seconds between a and b), and total heat experienced (shaded area under temperature curve between a and b in °s).

Biomass reduction due to prescribed burning was calculated as (1) the absolute difference between pre-burn mass and post-burn mass (g) and (2) the percentage of pre-burn mass removed (combustion completeness). Moisture content of the biomass was calculated as (1) the absolute difference between fresh and dry mass (g) and (2) the percentage moisture content as moisture content (g)/fresh mass (g). These values were calculated for total biomass and for the two main fractions (*Calluna* and litter) individually. *Calluna* and litter made up 99% of total dry biomass on average (Table 5.1), therefore the remaining fractions were not considered individually in these analyses.

The relationship between biomass reduction and other measured variables was investigated using linear mixed effect models (Pinheiro *et al*., 2009). Year, burn and thermocouple position were considered as nested random variables and the fixed effect variables were grouped into sets to build various maximal models. Percentages were arcsine square root transformed prior to analysis. Preliminary analyses revealed that models containing total pre-burn biomass (sum of *Calluna*, litter, moss and animal dung fractions) variables did not explain the biomass reduction data as well as those considering the two main fractions individually (*Calluna* and litter) based on AIC. Therefore, only analyses using *Calluna* and litter variables are presented here. From these variables, two sets were constructed. The first (A) used dry pre-burn biomass (g) and moisture content (g), and the second (B) fresh pre-burn biomass (g) and its percentage moisture content. This division was made to reduce redundancy of variables, i.e. inclusion of similar biomass measures. Maximum temperature, duration of burn and total heat, each at the base and top of vegetation, formed a third set of variables, hereafter referred to as burn variables. The two biomass variable sets, A and B, were each combined with the burn variables to give two maximal model specifications for subsequent simplification.

Table 5.1. Summary of all biomass and moisture content (MC) parameters from the 17 prescribed burns on Howden moor. Values for total biomass and for the two main fractions are given.

			n	Mean	SE	Maximum	Minimum
Total Biomass	Pre burn biomass (g)	Wet	68	567.6	27.3	1022.5	172.6
		Dry	68	300.8	14.4	538.0	94.0
	Pre burn MC (g)		68	266.8	14.0	522.2	78.6
	Post burn biomass (g)	Wet	68	174.8	15.1	528.4	31.3
		Dry	68	96.7	8.4	324.0	14.6
	Biomass loss (% pre burn mass)	Wet	68	66.9	2.6	93.3	3.5
		Dry	68	65.9	2.4	93.4	20.9
Calluna Biomass	Pre burn biomass (g)	Wet	68	310.3	14.7	672.1	79.3
		Dry	68	173.2	8.2	309.3	36.9
	Pre burn MC (g)		68	137.1	7.4	362.8	34.0
	Post burn biomass (g)	Wet	68	69.5	6.3	230.7	11.6
		Dry	68	42.2	3.8	132.7	9.5
	Biomass loss (% pre burn mass)	Wet	68	74.0	2.4	97.2	7.3
		Dry	68	71.4	2.6	96.7	16.6
Litter Biomass	Pre burn biomass (g)	Wet	68	249.3	18.4	689.4	53.4
		Dry	68	124.4	8.8	335.0	25.9
	Pre burn MC (g)		68	124.9	10.4	365.7	27.5
	Post burn biomass (g)	Wet	68	92.8	8.1	319.0	13.9
		Dry	68	51.2	4.6	200.8	9.4
	Biomass loss (% pre burn mass)	Wet	68	57.1	2.8	96.6	1.6
		Dry	68	54.5	2.8	95.9	0.2

Each specification was fitted to percentage total biomass reduction and simplified using forwards and backwards stepwise procedures based on AIC command stepAIC, package MASS: (Venables & Ripley, 2002). Minimum adequate models (MAMs) determined from each maximal model were compared using the AIC statistic to give an overall MAM (Crawley, 2007). The same procedure was followed using absolute total biomass reduction as the dependent variable and also percentage *Calluna* biomass reduction for comparison.

5.3. Results

5.3.1. Description of burn characteristics

Before burning the mean total dry biomass was 301g (173g Calluna and 124g litter; other fractions were negligible; Table 5.1). The maximum temperatures recorded in these 17 prescribed burns were 982°C and 993°C at the base and top of the vegetation respectively (Table 5.2). The burn residence time ranged from 11 to 547 seconds at the base of vegetation, with similar durations at the top (4 to 474 s). The burns removed up to 93% of total dry biomass, and up to 96% and 97% of litter and *Calluna* dry biomass respectively (Table 5.1).

Table 5.2. Summary of all calculated burn parameters from the 17 prescribed burns on Howden moor. The burn residence time was calculated between the first and last times at which temperature exceeded 60°C, and the heat experienced is the area integrated under the temperature curve between these two time points (Fig. 5.1).

		n	Mean	se	Minimum	Maximum
Maximum temperature (°C)	Base	53	662.3	31.3	78.6	981.5
	Top	53	705.5	25.5	269.5	992.8
Burn residence time (s)	Base	53	184.5	15.5	11.0	547.0
	Top	53	179.3	15.3	6.0	474.0
Heat experienced (° s)	Base	53	41525.4	3534.2	1168.8	124772.1
	Top	53	42673.2	3638.4	1837.9	125103.5

5.3.2. Factors influencing biomass reduction during prescribed burning

Aside from initial biomass (dry *Calluna* and litter), the only factor significantly influencing biomass reduction (percentage and absolute) during prescribed burning was burn residence time at the base of vegetation. Burn residence time at the top of vegetation was included in the maximal model but not the MAM. Therefore, unless otherwise stated, burn residence time hereafter refers to those measurements at the base of vegetation. Biomass loss was negatively related to burn residence time, i.e. more biomass was lost during shorter burns (percentage: Table 5.3a, Fig. 5.3a and absolute: Table 5.3b, Fig. 5.3b). *Calluna* biomass reduction was negatively related to maximum temperature at the top of vegetation as well as an effect of initial *Calluna* dry biomass (Table 5.3c, Fig. 5.3c). This was also the case for absolute *Calluna* biomass reduction, which is not further reported here.

Table 5.3. The Minimum Adequate Models derived from a linear mixed effects model for (a) biomass reduction (g) in relation to burn variables, and (b) the burn residence time (S) in relation to environmental predictors (here the intercept is Wind direction: east). The fitted estimate, standard error (SE), t value and P-value is provided for each term. The Deviance explained, and both AIC and DAC statistics are shown for both the MAM and the null model along with the % reduction (%Δ) in each. Significance of each term is coded as: + = P<0.10, **=P<0.01, ***=P<0.001.

(a) Biomass reduction (%)

Parameter	Intercept	Dry *Calluna* biomass	Burn residence time (base)
Estimate	0.832	0.001	-0.0005
SE	0.090	0.0004	0.0002
t	9.268	2.963	-2.251
P	<0.001***	0.006**	0.031+
	Null	MAM	%Δ
AIC	-8.127	-17.166	111
BIC	-0.246	-5.344	2072
Deviance	16.127	29.166	80.8

(b) Biomass reduction (g)

Parameter	Intercept	Dry *Calluna* biomass	Dry litter biomass	Burn residence time (base)
Estimate	-12.69	0.97	0.60	-0.17
SE	25.24	0.12	0.12	0.06
t	-0.50	7.81	5.17	-2.61
P	0.618	<0.001***	<0.001***	0.013+
	Null	MAM	%Δ	
AIC	643.84	575.01	10.7	
BIC	651.72	588.80	9.7	
Deviance	-635.84	-561.01	11.8	

(c) *Calluna* biomass reduction (%)

Parameter	Intercept	Dry *Calluna* biomass	Maximum temperature (top)
Estimate	0.99	0.001	-0.0003
SE	0.13	0.0004	0.0001
t	7.73	3.54	-2.10
P	<0.001***	0.001**	0.043+
	Null	MAM	%Δ
AIC	-11.16	-22.32	
BIC	-3.28	-10.50	
Deviance	19.16	34.32	79.1

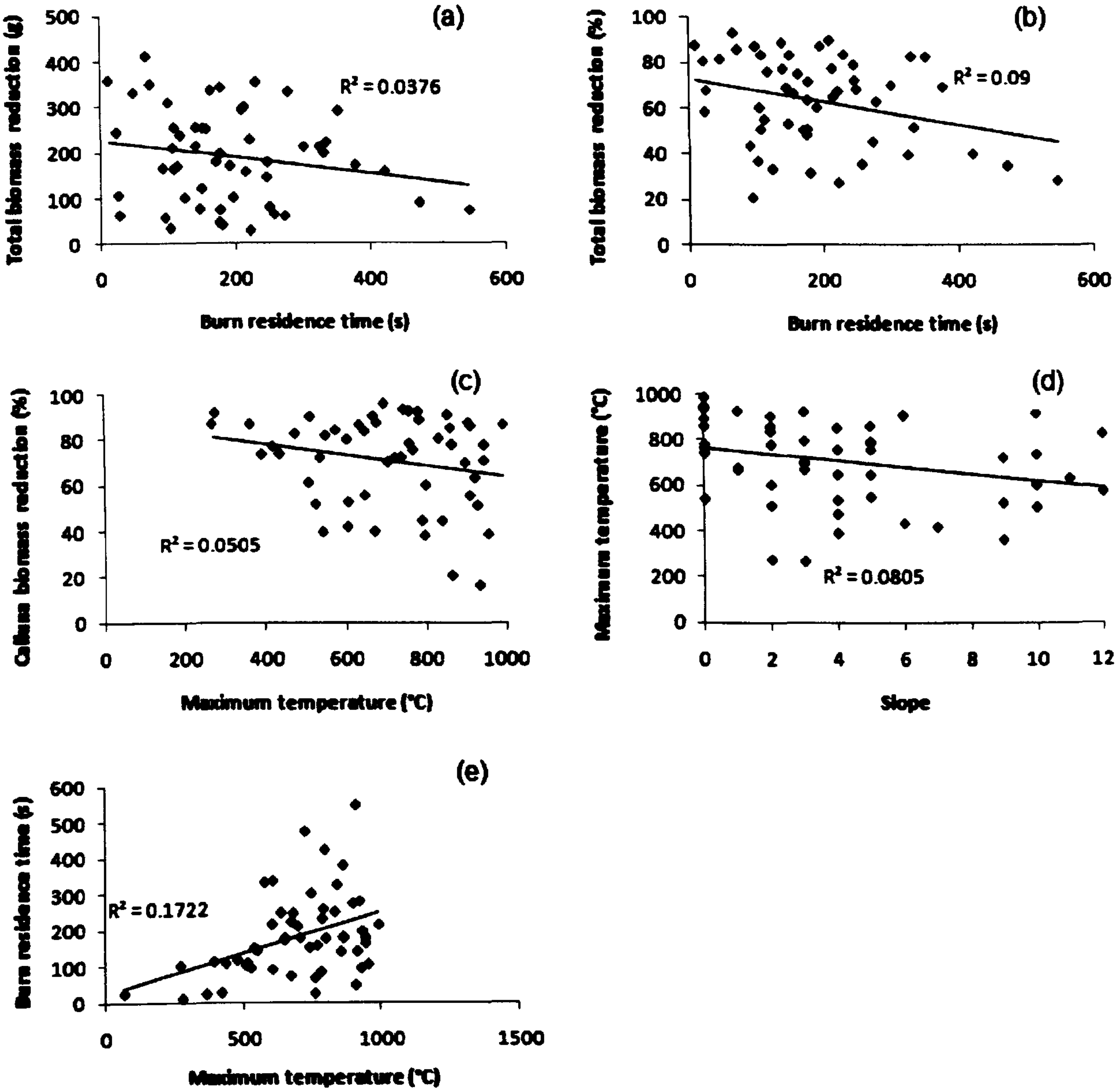

Fig. 5.3. The relationship between burn residence time and (a) total biomass reduction (g) at the base of vegetation (P = 0.164) and (b) total combustion completeness (%) at the base of vegetation (P = 0.0291). Relationships between (c) combustion completeness of *Calluna* and maximum temperature at top of vegetation (P = 0.106); (d) slope and maximum temperature (P = 0.0395); and (e) maximum temperature and burn residence time at the base of vegetation (P < 0.001).

5.4. Discussion

The aim of this experiment was to indentify the factors influencing biomass reduction in prescribed burns. A major result was the identification of burn residence time as the main "biomass loss predictor" variable. However, unexpectedly biomass reduction was greater in the shorter burns. This might be explained if shorter burns were hotter and therefore more intense (fast or "flash" fires), but burn residence time was positively correlated with maximum temperature (Fig. 5.3e) suggesting that maximum biomass loss occurs at low burn temperatures, although this did not feature in the MAM.

These conclusions are difficult to explain but some of the following factors are likely to be involved:

- Landscape and micro-meteorological factors with wind pattern varying during the burn and being affected by the wind. Where wind moves through the burn consistently a rapid fire consuming the vegetation and maintaining relatively low temperatures might be expected. Aspect and wind direction are likely to important variables here.
- The behaviour of moisture, both within and on the vegetation. If water droplets are evaporated during a fast fire then there may be negative feedback effects as the evaporation will reduce the air temperature to get over the high latent heat of evaporation of water. If this occurred in the fast burns then our temperatures recorded would be artificially low.
- In the slower burns, although they were measured as hotter there is a greater proportion of time at low temperatures where the vegetation is pyrolyzed to produce black carbon. Where this occurs there might be produced a protective layer which will impeded consumption by the fire. The charcoal remains will of course be included in the biomass recorded; whereas where complete consumption occurred it is converted to gas.
- It is possible that the two types of burn may be because in some situations the burn is sufficiently intense to initiate a "firestorm" situation in which the convection sucks in more air and becomes self-ventilating which leads to a positive feedback? Whether or not this occurred might depend on many factors such as meteorological conditions (and their history), topography (slope, shape), season, shape and size of the area of burning.

Further work is needed to elucidate the exact mechanisms between burning and fire severity *sensu* (Keeley, 2009). All of the fires reported here were carried out in early March into late April, and clearly information from burns over a longer period within the approved burning season and over a wider range of management regimes is also needed. Here, all of our burns were prescribed burns deliberately set to burn the vegetation as part of moorland management. None of the prescribed fires in this study burned into underlying peat, therefore this study only considers "light" severity burns sensu (Keeley, 2009; adapted from Ryan & Noste, 1985). This limits the conclusions that can be drawn, as more intense fires were excluded.

The PFA burning technique is based on the theory that lower temperatures ("cool burns") produce longer burn residence times, which remove less of the above ground biomass than a rapid hot burn would, as it leaves some of the bryophyte layer intact, albeit charred. The underlying peat is left undamaged. This, at least on this Peak District moorland, and under the conditions tested, supports this hypothesis in that slower PFA burns caused the least reduction in biomass. Shorter burns resulted in greater biomass removal, a removal the bryophyte layer is removed and peat may be exposed.

Chapter 6

Change in above-ground biomass, carbon and nitrogen occur during prescribed burning

6. Change in above-ground biomass, carbon and nitrogen occur during prescribed burning

6.1. Introduction

The use of prescribed burning has recently been reviewed (Tucker, 2003) and it is clear that moorlands must now provide a range of ecosystem services ranging from Biodiversity Action Plans through to the provision of carbon accounting the provision of clean water (MEA, 2005; Marrs *et al.*, 2007; Harris *et al.*, (in press). It is almost inevitable that burning will impact on water quality given that there will be a loss of "biotic control" *sensu* (Bormann & Likens, 1978) with an increased run-off and nutrient outputs. Here, the potential role of moorlands in the Peak District for carbon accumulation is considered. Fundamentally, prescribed burning will release carbon to the atmosphere, and then there will be accumulation of carbon as the vegetation recovers. The absolute amount of loss will depend on the biomass consumed by the fire and the ecosystem resilience, i.e. the time it takes for the ecosystem to recover (Mitchell *et al.*, 2000). The prescribed burn-cycle, therefore, reflects an initial loss followed by a gain and the overall budget will reflect these two processes. The aim should be to attain at least a balanced budget, i.e. zero loss. However, superimposed on this is wildfire risk, and here two factors need to be included in the discussion:

(a) A large biomass accumulation (fuel load) which will hold an increasing carbon store.

(b) Where there is wildfire then there may be an increased loss but also a variable ecosystem recovery, ranging from "as good as a prescribed fire" to very severe loss (Maltby *et al.*, 1990).

The aim of this part of this investigation is to start to provide information on biomass budgets during the prescribed burning cycle. A fundamental issue in this context is how the *Calluna* life-cycle operates in these upland moors. The *Calluna* life-cycle originally proposed by (Watt, 1947, 1955) was based on research done primarily based on work done at Lakenheath Warren on lowland heaths and on highly-exposed soils in mountainous areas, but not on vegetation on deep peat. Forrest (1971) suggested that at Moor House the moorland vegetation was at steady state but that the degenerate phase of *Calluna* was seldom seen and they hypothesised that the steady -state was assumed after 30 years free of burning and it was maintained by a complex set of factors including restriction of canopy height (presumably by climate and moisture conditions) upward growth of the bog surface the trailing growth form which presumably would allow layering of the *Calluna* stems and the gradual dying of old buried stems. These authors reviewed the literature and showed that the above-ground biomass varied between a mean above-ground summer biomass of 450-2930 gm^{-2} across Great Britain. Moreover, (Chapman, 1975), a lowland heathland in Dorset produced a model for above-ground biomass accumulation through time which produced an asymptote of 2500 gm^{-2} total above-ground biomass (2000 g m^{-2} for *Calluna*) after approximately 40 years. Clearly, the way that the vegetation biomass accumulates is fundamental to understanding the carbon cycle after prescribed burning. The aim of this paper is to assess the loss in above-ground biomass immediately after burning, and the subsequent recovery as the vegetation regenerates and aggrades (reorganization and aggrading phases, *sensu* (Bormann & Likens, 1978). Here, we report the loss in above-ground biomass and its associated carbon and nitrogen in 31 prescribed burns on Howden Moor in the Peak District and the accumulation of carbon in five separate moorland chronosequences after prescribed burning.

6.2. Methods

6.2.1. Experimental prescribed burns

A total of 31 prescribed fires implemented by the land manager as part of the normal burning regime on Howden Moor, Peak District, UK were studied (Table 6.1) between March and April in 2007, 2008 and 2009. Within each area to be burned, four measurement points were positioned in a standard 'zig-zag' pattern Harris *et al.* (2011). Before burning, the cover of plant species was estimated within a 50 x 50 cm quadrat at each measurement point. The vegetation was then removed to ground level using secateurs and sealed in airtight bags until weighed. A second quadrat, adjacent to the first was located in a random direction from the first. The plot was then burned by the land manager; a line of vegetation approximately 16-20m wide and at least 2 m upwind from the first sensor (Fig. 6.1) was sprayed with diesel and then lit with a flaming torch. The usual measures to control the fire were used; these included the use of fire beaters and fire foggers (http://www.firefighting.co.uk). Once cooled, vegetation from the secondary quadrat was removed and bagged as described above. Pre- and post-burn vegetation was separated into fractions (*Calluna*, litter, graminoids, bryophytes, and animal excrement) and weighed. Mass was recorded before and after drying in an oven until a constant mass was reached, thus giving fresh and dry mass values for each fraction; all samples were ground to pass a 1mm mesh. Total elemental C and N concentrations (%) were measured directly on randomly selected samples of each fraction collected pre-burn and post burn using a Carlo Erba Instruments NC2500 elemental analyser.

Biomass reduction due to prescribed burning was calculated as (1) the absolute difference between pre-burn dry mass and post-burn dry mass (g) and (2) the percentage of pre-burn mass removed (combustion completeness).

6.2.2. Chronosequence study: sites and sampling protocol

A space-for-time substitution study was carried out on five moorlands (Bamford, Broomhead, Howden, Midhope, and Snailsden Moors) in the North Peak ESA (Environmentally Sensitive Area), within the Peak District National Park, UK over a three-year period (Table 6.1). All of the burned patches on each Moor were mapped using aerial photography taken in September 2005 and cross-referenced with land burning management maps provided by the land managers. This provided a series of burned patches of known age (elapsed time since burning) between 2-16 years; older patches were also identified where no burning had been carried out for at least 35 years. There were 79, 249,103, 271, and 252 available burn patches on the five moors respectively. For the vegetation 10 different burns were selected using an age-stratified random sampling procedure from each Moor in each of the three survey years; the older patches were sampled in 2007 and 2008 (Table 6.1). Once selected, the geo-referenced outlines of each of the sampled patches were digitized from aerial photographs, and their areas calculated within (ArcGIS, 2009). The number of 1m^2 quadrats available within each burn patch was counted and a random selection made for field sampling (vegetation; n=10 in 2006 and 2008, n=4 in 2007, total n=1010).

Table 6.1. Details of the five study moorlands in the Peak District National Park, England where the post-fire vegetation succession was surveyed.

Moorland site	Longitude & Latitude	British National Grid squares digitized per Moor	Elevation range (m)	Age range (yrs)	Estimated age of older patches (yr)	Sampling years
Bamford	Latitude 53°21'N, Longitude1°40'W	SK 1993, 1994, 2093, 2094;	300-420	3-14	38	2006, 2007, 2008
Broomhead	Latitude 53°27'N, Longitude1°38'W	SK 2395, 2394, 2295 2294;	300-460	2-15	40	2007, 2008
Howden	Latitude 53°28'N, Longitude1°42'W	SK 2184, 2185, 2284, 2285;	272-540	2-15	50	2006, 2007, 2008
Midhope	Latitude 53°29'N, Longitude1°40'W	SK2198, 2197, 2099, 2098, 2097, 1999, 1998, 1997;	270-480	3-15	40	2007, 2008
Snailsden	Latitude 53°30'N, Longitude1°44'W	SE1503, 1501, 1404, 1401, 1400, 1304.	350-470	3-16	50	2007, 2008

On each of the five moors the patches and then the quadrat positions were located using GPS (eTrex Venture® HC). The cover of all higher plants, bryophytes and lichens, and a range of environmental variables (detailed in Chapters 2, 3). A vegetation sample was harvested from the central 0.25 m^2 sorted into five fractions (dwarf shrubs, graminoids, bryophytes, litter, and animal excrement), dried at 80°C and weighed. Biomass of each fraction was expressed in g m^{-2}. Formal quality control procedures were used at all stages of quadrat selection and in all aspects of the field work.

6.2.3. Data analysis

For the experimental prescribed burns, mean values of biomass before and after burning, and the % lost during the burn, were calculated for each of the 31 burns. To estimate the carbon and nitrogen loss, only the *Calluna* and litter fractions were included; this was justified on the basis that these fractions contributed more than 95% of the total biomass in the combined elapsed chronosequence (Fig. 6.1). The carbon and nitrogen content (g m^{-2}) was, therefore, estimated by summing the carbon/nitrogen proportions of each fraction multiplied by their respective biomass (Table 6.2). The carbon proportions were slightly greater in the post-burn samples, whereas there was little difference in litter concentrations.

Table 6.2. Carbon and nitrogen proportions (%) in *Calluna* and litter samples from both pre-burn and post-burn vegetation from selected burns on Howden moor, in the Peak District. Mean values (±SD) are presented.

Vegetation	Element	Pre-burn (n=27)	Post-burn (n=10)	t-value
Calluna	C	48.26±1.98	50.08±0.35	
	N	1.33±0.43	0.96±0.07	
Litter	C	48.97 ±2.67	50.39±0.18	
	N	1.42±0.45	1.44±0.05	

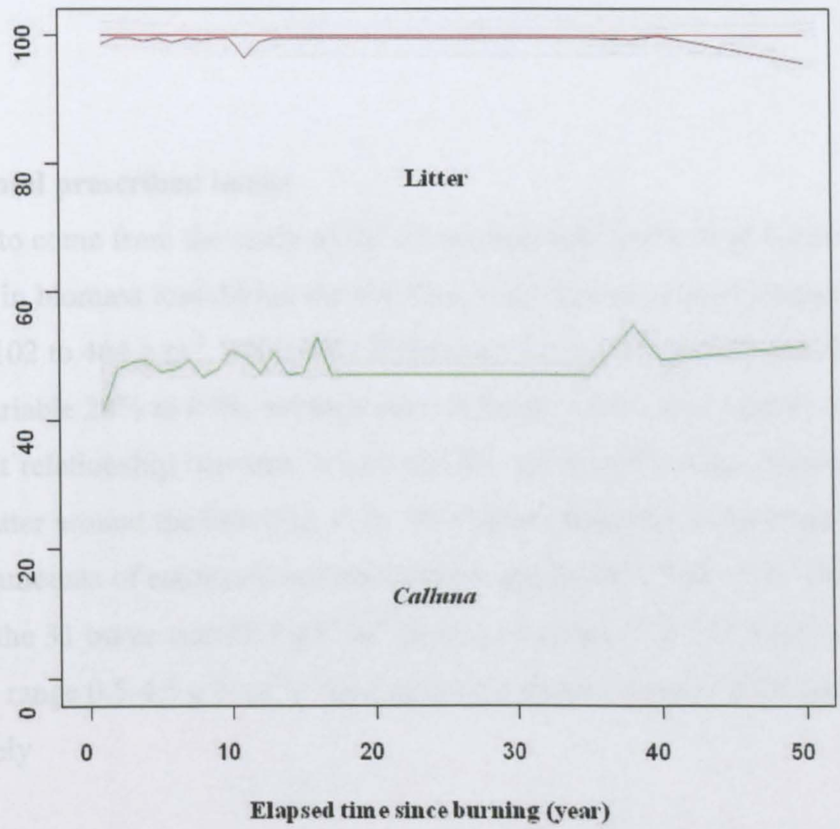

Fig. 6.1. Distribution (%) of the main fractions of biomass in moorland vegetation in chronosequences across five moors in the Peak District.

For assessing the relationship between biomass accumulation through time, initial inspection of the data showed a more or less linear relationship. However, when linear regressions were fitted between mean biomass per patch and elapsed time since burning inspection of residuals and Q-Q plots indicated non-normality and so log-log linear regressions and a range of non-linear equations were fitted (Crawley, 2007). In all analyses the reciprocal of the variance of each means was included a weight. The log-log fits are presented here as they provided the optimal balance between statistical fit and parsimony. Analysis of covariance was carried out relating biomass ($\log_e$) to elapsed time (log*e*) with Moor as a factor. Thereafter the model deletion approach was used to derive the Minimum Adequate Model (MAM), (Crawley, 2007) starting with the Moor x Elapsed time interaction model. The analysis was carried out for (a) the five moor survey and (b) Howden Moor where the mean biomass of the vegetation after the experimental burning (n=31 above) was combined with the chronosequence data from Howden.

6.3. Results

6.3.1 Experimental prescribed burns

The main result to come from the study of the 31 experimental prescribed burns was the very large variability in biomass loss during the fire (Fig. 6.2). The pre-burn biomass ranged from mean values of 102 to 464 g m^{-2}, a five-fold difference. The % loss of biomass during the fire was also very variable 28% to 86%; average over all burns = 62%; and median of 68%. There was a significant relationship between % loss and the pre-burn biomass, although there was considerable scatter around the line (Fig. 6.3). This large variability in biomass loss was also reflected in the amounts of estimated carbon and nitrogen losses (Table 6.3). The mean value calculated over the 31 burns was 89.9 g C m^{-2} (min-max range 12.7-155.3 g C m^{-2}) and 2.7 g N m^{-2} (min-max range 0.5-4.5 g N m^{-2}); this reflected a mean % loss of 59.0 and 64.0% for C and N respectively

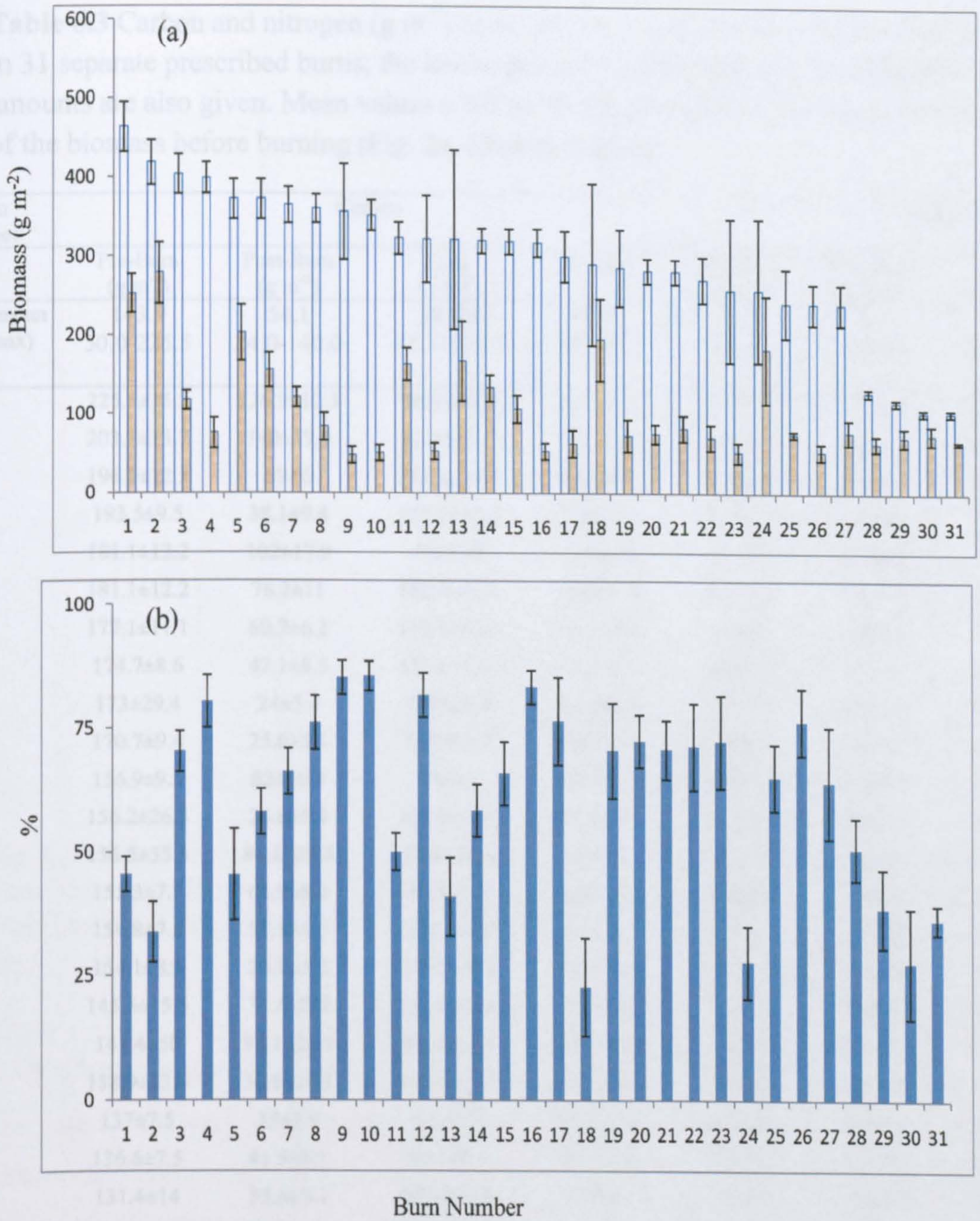

Fig. 6.2. Biomass lost after burning in 31 separate fires on Howden Moor in the Peak District, Derbyshire: (a) biomass (g m^{-2}) before (clear) and after burning (shaded); (b) % biomass lost during the fire. Mean values ± S.E are reported (n=4). Fires are plotted in rank order of the biomass before burning (a, clear bars)

Table 6.3 Carbon and nitrogen (g m^{-2}) in the above-ground biomass before and after burning in 31 separate prescribed burns; the loss expressed in g m^{-2} and as a % of the pre-burn amounts are also given. Mean values ± SE (n=4) are presented. Fires are plotted in rank order of the biomass before burning (Fig. 2a, clear histograms)

Burn number	Carbon				Nitrogen			
	Pre-Burn (g m^{-2})	Post-Burn (g m^{-2})	Loss (g m^{-2})	%Loss	Pre-Burn (g m^{-2})	Post-Burn (g m^{-2})	Loss (g m^{-2})	%Loss
Overall mean (min-max)	143.9 50.0-225.5	54.1 24.0-140.0	89.84 12.7-155.3	59.0 20.1-85.7	4.1 1.4-5.8	1.3 0.6-3.4	2.7 0.5-4.5	64.9 31.7-87.0
1	225.5±15.2	126.9±12.3	98.7±10.5	43.7±3.7	6.4±0.4	3.1±0.3	3.3±0.3	51.5±3.6
2	203.7±13.7	140±19.4	63.6±12.7	31.7±6.2	5.8±0.4	3.4±0.5	2.4±0.3	41.7±5.3
3	196.1±12.3	59±6	137.1±14.3	69.4±4.2	5.5±0.3	1.4±0.1	4.2±0.4	75±2.6
4	193.5±9.5	38.1±9.4	155.3±16.8	79.8±5.4	5.4±0.3	0.9±0.2	4.5±0.4	82.2±4.5
5	181.1±12.2	102±17.9	79±17.8	43.6±9.5	5.1±0.3	2.5±0.5	2.6±0.5	51.4±9
6	181.1±12.2	78.2±11	102.8±11.6	56.8±4.8	5.1±0.3	1.9±0.2	3.3±0.3	63.9±3.1
7	177.1±11.1	60.7±6.2	116.3±13.4	65.1±4.8	5±0.3	1.4±0.1	3.6±0.3	71.5±2.9
8	174.7±8.6	42.1±8.5	132.6±15.4	75.3±5.7	4.9±0.2	1±0.2	3.9±0.4	78.4±4.6
9	173±29.4	24±5.4	149±28.8	84.8±3.6	4.9±0.8	0.6±0.1	4.3±0.8	86.6±3.4
10	170.7±9.4	25.6±5.1	145.1±9.2	85.1±3.1	4.8±0.3	0.6±0.1	4.2±0.3	87±2.9
11	156.9±9.8	82±10.4	75±4.7	48.3±3.9	4.4±0.3	1.9±0.2	2.5±0.1	56.4±2.6
12	156.2±26.5	26.6±5.4	129.6±26.3	81.1±4.6	4.4±0.7	0.7±0.1	3.8±0.8	83.2±4.5
13	156.5±55.3	84.1±25.4	72.4±34.6	39±8.2	4.5±1.6	2.1±0.7	2.4±1	48.8±6
14	155.3±7.7	66.9±8.5	88.4±9.4	56.9±5.5	4.4±0.2	1.6±0.2	2.8±0.3	63.9±4.6
15	154.8±7.6	53.5±8.7	101.3±15	64.6±6.7	4.4±0.2	1.3±0.2	3.1±0.4	69.6±5.1
16	154.1±8.5	26.7±5.2	127.5±8.2	82.8±3.5	4.3±0.2	0.7±0.1	3.7±0.2	85±3.3
17	145.5±15.5	31.6±8.5	113.9±23.6	75.6±9.2	4.1±0.4	0.8±0.2	3.3±0.6	78.1±8.1
18	141.4±50	98.1±26.4	43.3±24.9	20.1±10.2	4±1.4	2.4±0.7	1.6±0.8	31.7±8
19	138.9±23.6	36.8±10.1	102.1±26.1	69.5±10	3.9±0.7	0.9±0.3	3±0.7	72.5±9.1
20	137±7.5	38±5.9	99±12.2	71.5±5.5	3.9±0.2	0.9±0.1	2.9±0.3	75.6±4.2
21	136.6±7.5	41.5±8.1	95.1±8.9	69.7±6.1	3.8±0.2	1.0±0.2	2.8±0.2	73.7±5.7
22	131.4±14	35.6±8.1	95.8±20.8	70.3±9.2	3.7±0.4	0.9±0.2	2.8±0.5	73.7±7.6
23	125.7±44.4	26.1±6.5	99.5±41.4	71.5±9.7	3.6±1.3	0.6±0.2	2.9±1.2	76.1±7.9
24	125.7±44.4	92.1±34.4	33.6±18.3	25.4±7.5	3.6±1.3	2.2±0.9	1.4±0.6	37.5±7.6
25	117.4±21.3	37.9±1.9	79.5±21.6	64.0±6.9	3.3±0.6	1.0±0	2.3±0.6	67.2±6.5
26	116.8±12.4	26.7±5.2	90.1±15.1	75.6±7	3.3±0.3	0.7±0.1	2.6±0.4	78.5±6.5
27	116.4±12.4	39.2±7.7	77.2±19.9	62.8±11.7	3.3±0.3	0.9±0.2	2.3±0.5	68.1±9.4
28	62.5±2.3	32.2±5	30.2±3.0	49.0±6.4	1.8±0.1	0.8±0.1	1±0.1	56.9±4.6
29	56.4±2.1	36.2±5.9	20.3±4.4	36.6±8.2	1.6±0.1	0.9±0.1	0.7±0.1	45.9±5.5
30	50.2±1.9	37.4±5.9	12.7±6.1	25.1±11.4	1.4±0.1	0.9±0.1	0.5±0.1	36.2±9
31	50.0±1.8	32.8±1.0	17.2±2.0	34.2±3	1.4±0.1	0.8±0	0.6±0.1	43.3±4.7

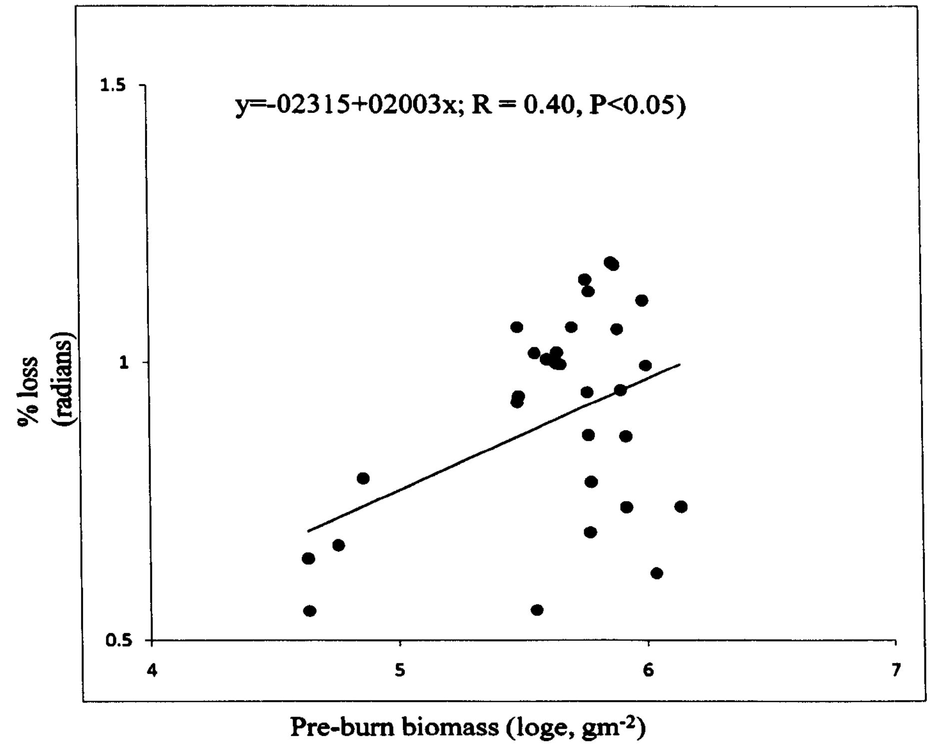

Fig. 6.3. Relationship between mean biomass loss (% arcsine transformed to radians) and mean pre-burn biomass ($\log_e$, g m^{-2}) in 31 separate prescribed burns on Howden Moor in the Peak District, Derbyshire; mean values (n=4) are plotted.

6.3.2. Chronosequence study

The data for the very oldest stands in these five moors on the Peak District were considerably greater than comparable data from the literature, 5240-10,000 g m^{-2} compared with literature values of 450-2000 g m^{-2} (Table 6.3). However, the oldest stands studied here were at least 20 years older were 8-15 years older than nearest equivalent aged stand at Moor House and most other studies. The younger stands studied were within the literature ranges.

The MAM for the ancova analysis of the entire chronosequence data from all five moors selected the maximal model with an adjusted R^2=0.9694. Assessing the individual coefficients from this model (Table 6.4) showed that two moors Broomhead, Snailsden were not significantly different from the intercept Bamford, whilst Howden and Midhope showed a significant Moor effect, both having a greater initial biomass than the intercept. There was a significant relationship with elapsed time since burning (Fig. 6.4), but only Howden showed a significantly different slope from the intercept. All moors showed a consistent increase in biomass with time over the time span available in this study when expressed on a log-log basis. The repeated analysis for the combined data from Howden (Fig. 6.5) showed a similar log-log relationship; the data for all 31 prescribed burns (Fig. 6.5a) show a considerable spread but are still within the expected ranges.

Table 6.4 Comparison of above-ground biomass data (g m^{-2}) from the oldest stands on five moors from this study (minimum and maximum means ± SE of individual patches) and literature values.

Location	Site	Elevation (m)	Stand Age (years)	Biomass (gm^{-2})	Reference
Peak District	Bamford	300-420	38	5364±569-8019±270	This study
Peak District	Broomhead	300-460	40-50	6540±844-7114±701	This study
Peak District	Howden	272-540	50	5577±470-6401±596	This study
Peak District	Midhope	270-480	40	5241±660-10024±337	This study
Peak District	Snailsden	350-470	40	5250±662-6507±549	This study
Moor House	Rough Sike	305	>30	*Calluna* 970 Total 1300	Forrest (1971)
Moor House	?	?	?	450-600	Allen (1964)
Moor House	?	?	?	790	Gore & Olsen (1967)
Hexham	Blanchard moor	305	10	1920	Robertson & Davies (1965)
Kincardineshire	North Cairn o' Mount	274	15	2930	Robertson & Davies (1965)
Kincardineshire	Kerloch Moor	140-280	25	1840	Kayll (1966)
Teesdale	?	290-850	6	600	Bellamy & Holland (1966)
Teesdale	?	290-850	14	2000	Bellamy & Holland (1966)
Dorset	Poole Basin	90	?	1820	Chapman (1967)
Dartmoor		320-340	?	2000	Chapman (1967)

Table 6.6 Minimum Adequate Model from the ancova testing the relationship between biomass through time (ET) at five separate moors in the Peak District. Significance: ** = P<0.01, ***= P<0.001. Bamford was the intercept.

	Estimate	Standard Error	t value	P	Significance
Intercept	4.9533	0.04872	101.677	<0.001	***
Broomhead	-0.11278	0.06613	-1.705	0.090503	-
Howden	0.39148	0.08824	4.437	<0.001	***
Midhope	0.22997	0.08332	2.76	0.006611	**
Snailsden	0.17888	0.12553	1.425	0.15654	-
ET	0.99452	0.02941	33.821	<0.001	***
Broomhead x ET	0.02329	0.04064	0.573	0.567588	-
Howden x ET	-0.18208	0.04731	-3.848	0.000186	***
Midhope x ET	-0.02116	0.04988	-0.424	0.672122	-
Snailsden x ET	-0.06168	0.06183	-0.998	0.320283	-

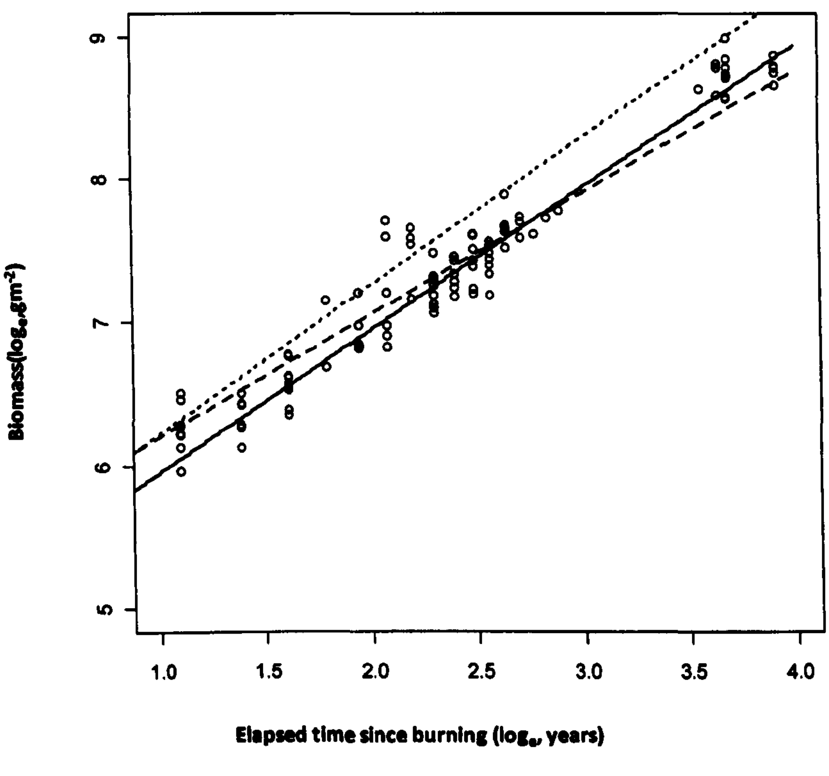

Fig. 6.4. Relationship between mean biomass (loge, g m^{-2}) and elapsed time since burning on five moors in the Peak District. Regression lines from ancova (Table 6.2) are plotted along with mean values. Bamford, Broomhead and Snailsden (solid line), Howden (dashed line), Midhope(dotted line).

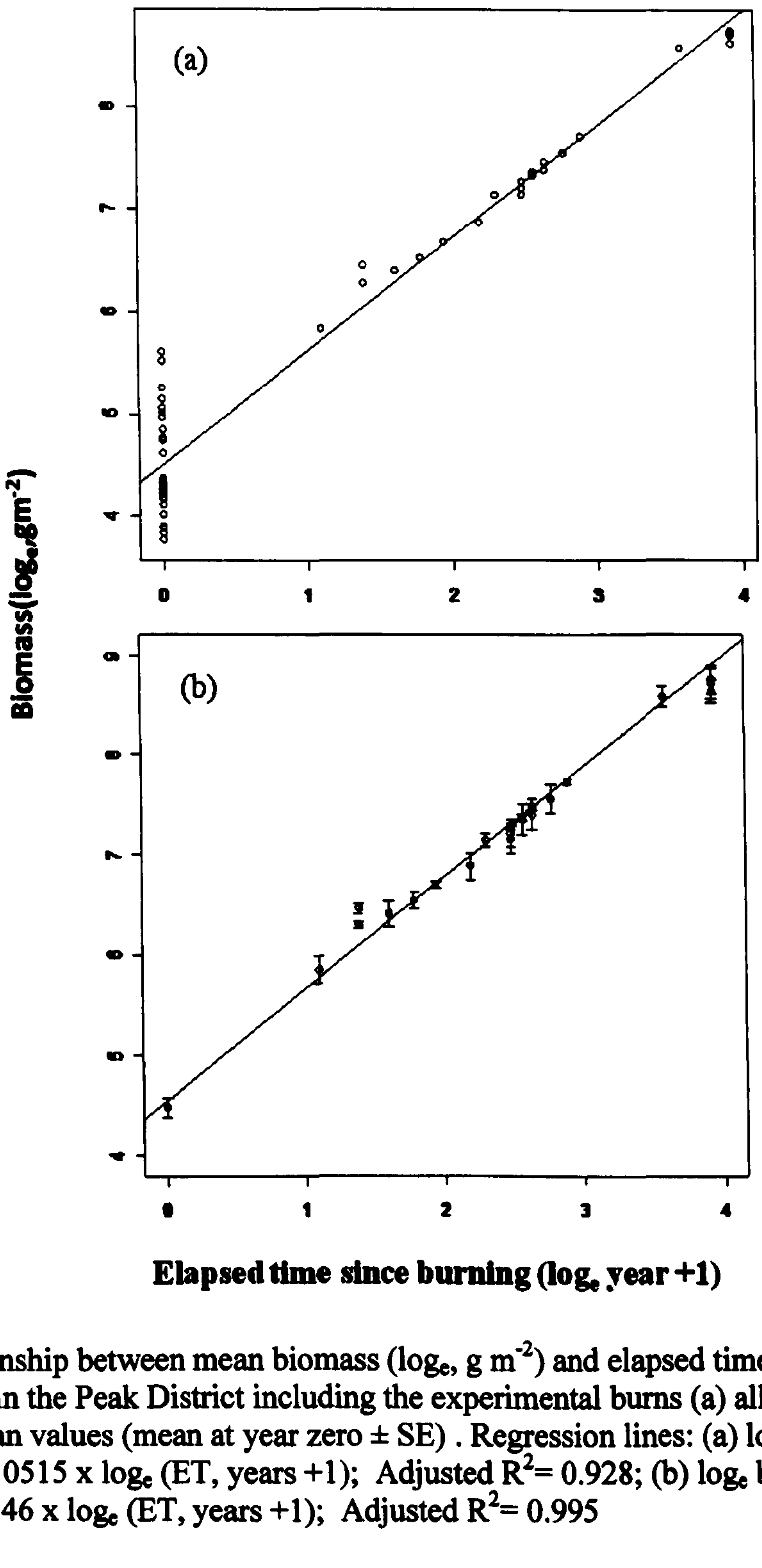

Fig. 6.5 Relationship between mean biomass ($\log_e$, g m^{-2}) and elapsed time since burning at Howden Moor in the Peak District including the experimental burns (a) all burns (mean values), (b) mean values (mean at year zero ± SE) . Regression lines: (a) $\log_e$ biomass (g m^{-2}) = 4.49981 + 1.10515 x $\log_e$ (ET, years +1); Adjusted R^2= 0.928; (b) $\log_e$ biomass (g m^{-2}) = 4.53796 +1.11146 x $\log_e$ (ET, years +1); Adjusted R^2= 0.995

6.4. Discussion

The data collected here provides comprehensive information on biomass budgets during the prescribed burning cycle. However, central of this issue is that these moorlands do not appear to be following the "traditional *Calluna* life-cycle model" in that the degenerate phase does not appear to be pronounced, and in the oldest stand there is new *Calluna* growth produced by layering shoots (McDonald *et al.*, 1995). This is in keeping with (Forrest's, 1971) observations that at Moor House the moorland vegetation was in a presumed steady state but the degenerate phase of *Calluna* was seldom seen. From the data presented here it is hypothesised that the moorland vegetation continues to grow and layer at least over the 50-year period studies here, and (a) there is no evidence to support this view that the *Calluna* will become degenerate and allow other species to colonize beyond this period, and (b) it is possible that the large litter layer accumulated (~50%) acts as an inhibitory factor that reduces species productivity *sensu* (Connell & Slatyter, 1977). In order to resolve this problem clearly studies on older aged moorland in the Peak District are need. Unfortunately these moors do not exist because of past burning practice (prescribed burning and wildfire combined) but the data presented here provides an excellent baseline for future studies.

6.4.1. Loss of biomass, carbon and nitrogen during prescribed burning

One of the surprising results from this study was the very large variation in biomass (and predicted carbon and nitrogen loss) after prescribed burning. Data from 31 separate burns were available The pre-burn biomass range to some extent reflected the policy of the moorland manager for burning individual patches within his burning management plan, and the range from 102 to 464 g m^{-2} is at the lower end of the biomass values detected across all five moors. However, the variation in % loss of biomass during the fire average 62% but with a wide spread 28% to 86%. Clearly the fire conditions can significantly affect biomass loss. The PFA-burning approach is designed to minimize biomass loss and maintain as much *Calluna* stick and bryophytes as possible, In some instances this appears to be working, but where large biomass losses occur then it has been less successful The reasons for the variation in burning performance remain unknown but will almost certainly involve meteorological conditions before and during the fire, the fuel load and the its moisture content, the wind speed and the skill of the burning operators (Chapter 5). Estimates of the potential loss of carbon and nitrogen during the burns were made and this averaged 89.9 g C m^{-2} (min-max range 12.7-155.3 g C m^{-2}) and 2.7 g N m^{-2} (min-max range 0.5-4.5 g N m^{-2}).

6.4.2. Biomass accumulation after prescribed burning

After prescribed burning the ecosystem recovers from a combination of resprouting and the colonization of new species. The aim of prescribed burning is to minimize damage and maximise recovery, i.e. minimize ecosystem damage *sensu* (Keeley, 2009) and hopefully develop a system that has high resilience (Mitchell *et al.*, 2000). This is effectively the "reorganization, aggrading and steady-state phases" as described by (Bormann & Likens, 1978). The major result in this study was that the oldest stands had a much greater biomass than reported in other presumed, albeit younger, "steady-state" ecosystems. Here, there was no evidence found to support the view that the system was approaching steady-state; the relationship between biomass accumulation with time was best fitted with a log-log regression and whilst this relationship does suggest a curvilinear relationship, it has not approached an asymptote over the period studied. It is always problematic to predict beyond the limits of the data collected, but fitting a Lineweaver–Burk plot to these data suggest predict an asymptote of ca, 15,000 g m^{-2} at ca. 115 years. What these data and this analysis shows is that information on the long-term accumulation of biomass on these moorlands is uncertain, and assumptions of steady-state in other moorland studies may be optimistic. These data do suffer for a lack of long-term data but there is also a gap in the 20-40 year period which would also help to improve accuracy in developing a predictive relationship; however, unfortunately stands within this age range do not exist. Again the information collected here provides a baseline which can be used in future studies to improve our knowledge and understanding.

6.4.2. Future studies

Prescribed burning does release carbon to the environment and here a first approximation has been made to assess the amount and variability of the amounts released. Carbon is also taken up. The vegetation and in the very old stands there were biomass values between 5,000-10,000 g m^{-2}. At 48% carbon and 1.35% nitrogen (Table 6.2) this translates to a carbon stock of 2,400-4,800 g C m^{-2} and

67.5-135 g N m^{-2}. Both of these figures are much greater than the amounts lost during prescribed burning (89.9 g C m^{-2} and 2.7 g N m^{-2}). Clearly, from a carbon and nutrient accounting point of view the no burn option is to be preferred. However, a no-burn option allows a very large fuel load to develop and hence will provide very large quantities of carbon and nitrogen to be lost in a single wildfire. As such a wildfire could be extremely damaging then the percentage losses produced here from prescribed burns might be very much greater and approach 100% of the above-ground biomass and may even remove the peat surface layer also (Peak District National Park, 2006). The data collected here will be used in future modelling studies to develop optimal burning management plans that produce biodiversity outcomes and minimize wildfire risk. Here, only the above-ground biomass has been studies, and it is accepted that this is only part of the biomass, carbon and nitrogen cycles. Transfer to peat and root growth has been ignored. Further studies are needed to expand our initial findings.

Chapter 7

General Discussion:
The future of British moors and the role of prescribed burning

7. General Discussion

Within the UK most heathlands and moorlands are a palimpsest in that they reflect the development of ecological communities and the impact of human societies since the last Ice Age (Pearsall, 1950; Gimingham,, 1972). The current vegetation in most upland areas results from the management that has been applied over the last two hundred years or so and for the most part this has been for sheep grazing and grouse shooting (Chapter 1). In recent years there has been a considerable increase in interest in upland moorlands, to some extent started by the "Mass Trespass" on Kinder Scout in the 1930s. This trespass was organized in response to the restricted access to walkers enforced on the moors by owners, and the wish for walkers to roam freely. After the Second World War, the creation of National Parks in England And Wales, including the Peak District NP, which was the first one created (PDNPA, 1958) and more recently the impact of the CROW Act (Countryside and Rights of Way Act) (CROW Act 2000; JNCC, 2000), have increased the rights of access and, therefore, the number of people using the moors. Recently the creation of an NGO (Moor for the Future) has galvanized activity on moorland restoration, knowledge exchange and promotion of moorlands as important landscapes (Bonn *et al.*, 2009). Thus, the objectives for moorlands are of interest to a wide variety of stakeholders (Goldthorpe *et al.*, 1963).

The determination of management objectives for any moorland must reflect the requirements of the owner, which can be private individuals or consortia, NGO or Utility Company, who often lease some of grazing and shooting rights to tenants. The recent emphasis given to ecosystem services on moorlands for both provision of clean water and carbon sequestration may alter the balance of management once economic values are put on these services. In all moorlands there should be a move towards maintenance of existing high quality habitats, and restoring ones that have been degraded through time. Such degradation includes (a) reduction the species pool (Chapter 2), (b) increased gulley erosion which leads to a lowering of water tables, increased DOC and POC release into waterways, hence reducing drinking water quality. The threats to the future management of the moors will include (Table 7.1):

- Global climate change. Increasing temperatures and summer drought will dry out the surface organic soils, reduce the peat depth and shift the balance to species of drier communities. The effects of climate change on the ecosystems of the British uplands have recently been reviewed (House et al., 2010) and they argue that the uplands are likely to experience a change in climate away from that currently associated with upland environments, peat growth and especially *Sphagnum*. They predicted the vegetation will change to species typical of drier communities and would contain more grasses, and that the ecosystems would be subject to more intense erosion, especially where the water table had already been reduced. These changes would add to those currently imposed by management and land use impacts.

- Diffuse atmospheric and local point-source pollution (Tallis, 1964; Holden *et al.,* 2006; McDonald *et al.*, 2000),
- Direct loss where land is changed from moorland to construction or afforestation (e.g. Webb, 1986), the development of moorland vegetation after trees are harvested (Gimingham, 1972), (d) agricultural changes including changed grazing management (e.g. Palmer *et al.*, 2004) and (d) wildfire (Maltby *et al.*, 1990, 2009).
- The impact of agriculture has been considerable over the last three centuries with a reduction in cattle grazing the moor, and a switch to large-scale sheep grazing (Hester, 1992) which led to a large shift in vegetation from moorland to grassland. In the latter part of the twentieth century, steps were taken under agri-environment schemes to reduced sheep numbers and hopefully start to improve these upland ecosystems (Hester, 1992).
- Many upland moors are managed for sporting purposes, and this usually involves the use of prescribed burning. Prescribed burning has been considered damaging to some users, e.g. water companies (Yallop *et al.*, 2006), where water discoloration, found on some moors after burning is costly to remove (Worrall *et al.*, 2010). It is also inevitable that moorlands will contribute to carbon credits because of the large carbon store held in the organic matter of peat soils (Clay *et al.*, 2009).
- Moorland management has also attracted adverse criticism from some conservation groups and the media through association with the practice of predator control, which impinges on raptor populations on moorlands (Redpath *et al.*, 2010). As raptor persecution is illegal in the UK this should not be considered part of any moorland management.
- In addition, the current vegetation of upland moors is adapted to fire management, with the dominant species being stimulated by fire and or smoke; they are, therefore prone to wildfire; most usually such fires are set off accidentally or through deliberate wildfire.

Set against this background of known environmental pressures of the uplands moors of the UK, there is a need to comply with the UK government's response to EU legislation. There is a requirement as part of the UK's Biodiversity Action Plan strategy to survey habitats, assess their condition, and where possible increase the amount of each habitat into favourable status (Anon, 2005). For upland heath the assessment reported in 2006 showed that almost half the upland heath and three-quarters of blanket bog in SsSSI were in "Favourable" condition or "Unfavourable recovering"; for Natura 2000 sites the figures were slightly lower, 40% and 59% respectively (Fig. 7.1, Williams, 2006). The factors that indicated

"Unfavourable conditions" (Fig. 7.1) included over-grazing and burning as the top two. However, in many upland heaths and blanket bog it is not clear whether evidence of prescribed burning was included as a damaging feature. Irrespective, the presumption is that now blanket bog will not be burned (Glaves *et al.*, 2006; Worrall *et al.*, 2010).

Table 7.1. A summary of environmental pressures affecting ecosystems in the uplands of the United Kingdom.

Indirect pressure: large-scale	**Direct pressures**	**Local pressures**
• Atmospheric pollution deposition • Climate change	**Infrastructure** • Construction • Road building • Waste management • Soil erosion and land sliding • Mining (new and legacy) • Hydro-electric • Wind farms **Forestry** • Afforestation • Deforestation including logging and forest fires • Forest management activities	**Land use management/change** **Agricultural:** • Agriculture/Livestock • Abandonment • Improving of grasslands • Heavy Stocking (overgrazing, increases tracks, burning) • Improving animal production • Change in the type of grazing • Specialisation (over dominance of sheep) **Game management** • Illegal persecution of raptors • Introduction of birds • Poaching and culling **Access and Recreation** • Access • Sports • Recreation • Hunting **Wildfire**

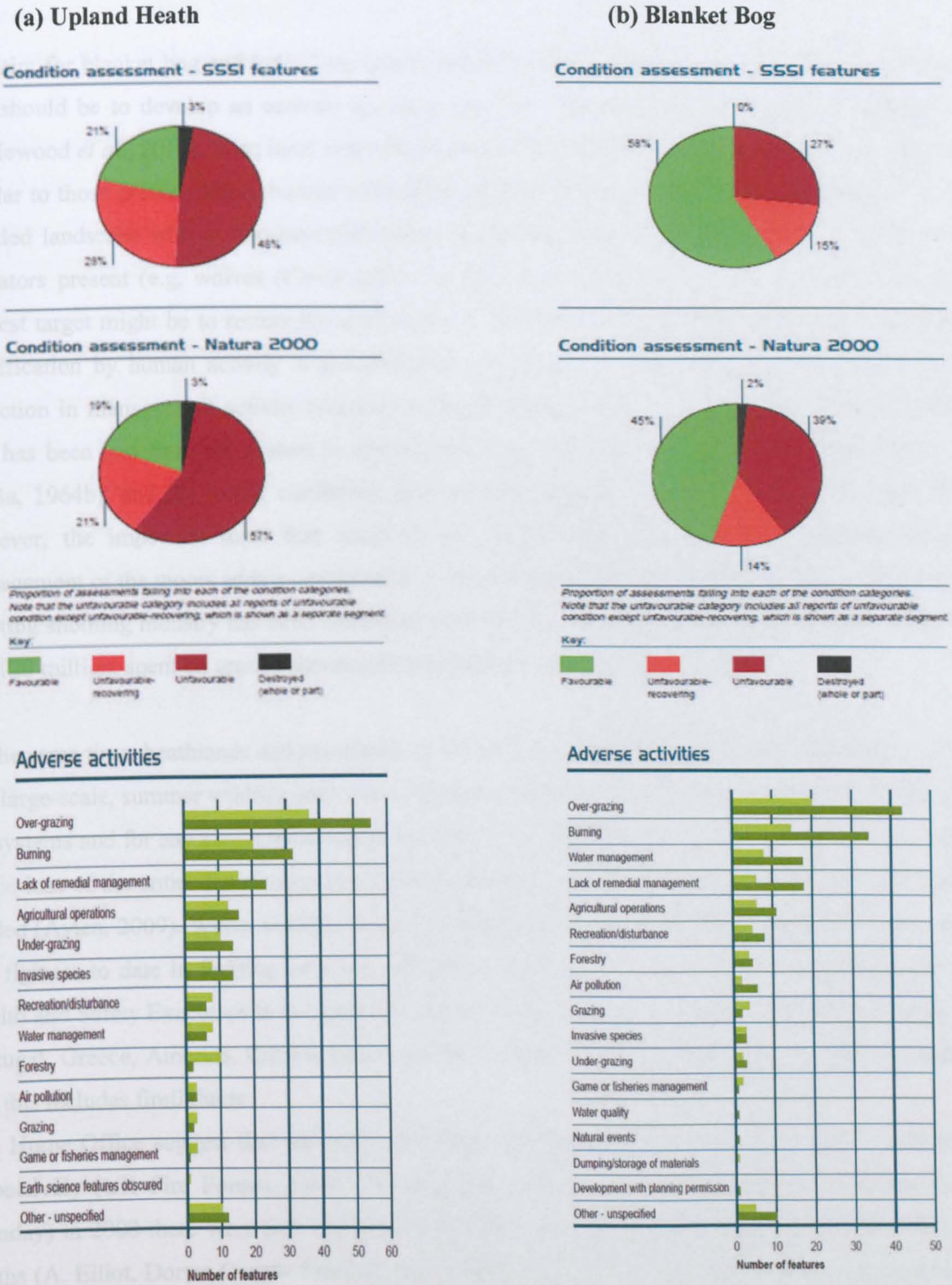

Fig. 7.1. The status of (a) upland heath and (b) blanket bog in upland UK along with an assessment of the features that rendered sites unfavourable (Williams, 2006). Data for sites designated as SSSIs and Natura 2000 are provided; there are some sites in common.

7.1. The restoration strategy for upland moorland in the UK

The aim for blanket bog and indeed any peaty soil should be to maintain the carbon store. For peat, the aim should be to develop an actively growing bog with well-developed acrotelm and catotelm layers (Littlewood *et al.*, 2011). In an ideal world restoration should have as a target a "pristine" system, i.e. one similar to those present before human influence modified the system. Thus, the target might be a lightly wooded landscape with a moorland understory, grazed by large herbivores such as Red deer with top predators present (e.g. wolves (*Canis lupus* L.), lynx (*Lynx lynx* L.), bear (*Ursus arctos* L.). A more modest target might be to restore the ecosystems to a system present in more recent times, where some modification by human activity is accepted but this impact is relatively small. This would involve a reduction in management activity (grazing, managed shooting) and a re-instatement of (1) biodiversity that has been lost from the system in recent times (e.g. bryophyte species in the Peak District, Tallis, 1964a, 1964b), and (2) wetter conditions than occur at present to encourage peat formation. There is however, the important issue that much of the land is under private ownership and the current management of the moors adds a considerable economic input into the local upland economies. The larger sporting shooting industry has been calculated to be worth £1.6 billion to the UK economy, with 13.33% or £120 million, spent on grouse-shooting in good grouse years (PACEC, 2006).

At the same time, heathlands and moorlands in the UK are a potential wildfire risk (Chapters 1, 5, 6) and any large-scale, summer wildfire could have devastating consequences for the continuation of the existing ecosystems and for any ecosystems restored in the future (Maltby *et al.*, 1990). Such wildfires could lead to the loss of the entire functioning ecosystem and after very severe wildfires, metres of peat have been eroded (Aylen, 2009). Whilst wildfire in the UK is a problem financially there has been no loss of life of fire fighters to date in fighting wildfires. However, there have been examples of investigations by the Health and Safety Executive in Scotland (M. Bruce, 2005) and in other parts of the world (France, Spain, Portugal, Greece, America, Canada and Australia, Catry *et al.*, 2010) there is loss of life in most years, and this includes firefighters.

The Home Office suggest that reported arson fires increased 8.4% from 2007 to 2009, 2,198 to 2,400 respectively (UK Fire Forum, 2009). For example, over the Easter holidays (Good Friday to Easter Monday) in 2008 there were 547 call outs by the Fire Rescue services to reported arsons on the Dorset heaths (A. Elliot, Dorset County Council, pers. comm.; UK Fire Forum 2009). These call outs represent the total number logged and it is not clear whether some of these were multiple callout or not. Nevertheless, it is a large number of incidents in a four day period. This willful fire rising on moorland has resulted in prosecutions. In 2009, three men were fined a £1,200 for damaging an internationallyprotected site, by deliberately setting fire to a moorland area within the Peak District National Park (Natural England, 2009). The incident happened on land owned by United Utilities adjacent to the Holme Moss road within the Dark Peak Site of Special Scientific Interest (SSSI) on 8 June 2008.

The Fire and Rescue Service (FRS) in the UK categorize fires on the basis of the size and time it takes to control them: (1) small '4-hour' fires of typically less than 15 ha, and (2) large' 3-day' fires of over 350 ha. From a preliminary study of fourteen fires, the '4-hour' fires cost FRS ca. £15,000, and the '3-day fires' £210,000 (Peak District National Park Authority, 2009). Such figures exclude helicopter costs, damage to property, livelihoods and other ecosystem services, or injuries to FRS officers. Most fires are reported in the late afternoon, so ideally the fire should be tackled hard and quickly before FRS officers are stood down in the evening, to prevent the fire developing overnight into a much more costly '3-day' fire. Costs of restoration after wildfire are prohibitive; it has cost £1.25m to restore 4.3 km^2 of peatland at Bleaklow (Moors for the Future, 2009). Approximately another 3 km^2 of the area burned in the April 2003 fire remains to be restored; the estimated costs for this are £870,700. These are direct costs and do not include the overheads of the NGO who have carried of the restoration work (Moors for the Future). It has been estimated that if a fire can be contained as a '4-hour' fire it will not only save money but also save 335 ha of ecosystem damage. An approximation to the cost breakdown of a single wildfire on Bleaklow is presented in Table 7.2.

Table 7.2. Approximate costs of fighting the Bleaklow Fire in 2003 (Peak District National Park, 2009).

Duration	**5 days** Broke out on Kinder Scout	Start: 13.48, 17th End: 18.08, 22nd
Area burnt	**744 hectares** (burns into peat) Water catchment for United Utilities	23.9°C maximum temperature on 17th, day before Easter; winds 4.0,3.6,5.8,4.9,1.8,2.7 ms^{-1} easterly, dry
Total appliance hours	**450 hours** Pumps + ATVs + fogging units + hoses	£1,000 PER/HR
Apportioning costs	**Fire Rescue Service** £450,000	Helicopter call-out Civilian and RAF £55,000 Fencing etc £45,000
Total cost		£550,000

How then should restoration proceed? There is some considerable debate as to the optimal way to achieve conservation of many upland habitats, and especially blanket bog. It is accepted that these habitats have been modified considerably by man's activities and they are much drier than they were in the past. There are essentially three schools of thought for the management of moorlands, and these can be briefly summarized as follows:

(1) The "do-nothing" option; here the aim is to allow the dominant *Calluna* plants to continue to grow and to follow the traditional Watt cycle (Watt, 1947, Fig. 1.2), and when the *Calluna* enters the degenerate phase other species will colonize. Moreover, the hope would be that as the *Calluna* grows in height, the bryophytes grow up with it and maintain a presence throughout the life-cycle. The evidence presented here suggests that in the Peak District moorlands the Watt cycle (Fig. 1.2), if it does operate at all, operates at much longer time-scales than have been reported elsewhere (Chapters 2, 3, 5, 6). Here the evidence shows (a) the only species to increase continuously in the 50-year post-burn succession is *Calluna*; all others show evidence of a unimodal peak before the 50 years, and (b) the major components of the vegetation are mostly live *Calluna* and its litter. Essentially, without burning the vegetation reduces in plant diversity and creates a monoculture of *Calluna* and its litter. There was no evidence to suggest that under the current conditions on which these moorlands persist that either late-successional species were invading or bog-forming species were growing up with the *Calluna* as it increased in height.

   With respect to wildfire, the "do nothing" option the in Peak District would produce a standing biomass estimated at 15,000 g m^{-2} at equilibrium and even after 50 years there is a mass of ca. 7,000 g m^{-2} (Chapter 6). This biomass, especially if burned under hot, dry summer conditions, is predicted to produce very intense fires which would be very damaging. This prediction has not been tested as it would be illegal to light experimental fires under such conditions. Nevertheless, with the high density of tourist in the Peak District this wildfire risk cannot be ignored.

(2) The "wet-up" option, where the moisture holding capacity for the moorlands is increased by blocking water run-off through gulleys (Worrall *et al.*, 2007), this technique known as gulley-blocking has been tested experimentally in some places and success has been variable (Worrall *et al.*, 2003). This technique is hypothesized to work by reducing the growth of *Calluna* as the site becomes wetter and allowing other peat-forming species (e.g. *Sphagnum*) to colonize and take over. This may occur in some places and if it does occur there will be an increase in carbon sequestration as the peat develops. However, the increase in reducing conditions might enhance the volatilization of methane, a greenhouse gas (Holden *et al.*, 2006). There is also the intriguing

question of how much carbon has been lost to CO_2 as a result of peat drying out in the past. Unfortunately, a greenhouse gas balance sheet has not been calculated for moorland areas where gulleys have been blocked and no has an attempt been made to assess losses through past drying out.

With this option, it might be assumed that wildfire would not be a problem as the sites should be much wetter and this should have a negative feedback effect on *Calluna* growth forcing a change to an ecosystem with a greater bryophyte and graminoid component, a much lower biomass (fuel load) and peat development. This strategy may work but the surface vegetation might still be affected by wildfire in very dry years, the frequency and intensity of which is predicted to increase with global climate change (House *et al.*, 2010).

Fig. 7.2. A view on an area of Kinder Scout where gulleys have been blocked and the areas are beginning to regenerate (photo by G Eyre 2008).

(3) The "managed" option, where the existing vegetation is managed to maintain at least some diversity, albeit it may be a restricted sub-set of the potential biodiversity that could be present, and where the vegetation biomass (fuel load) is minimized. The most common approach to this form of management is prescribed burning (the topic of this thesis), but some practitioners favour the use of cutting as an alternative. There is very little experimental data on the response of vegetation to cutting; the only published comparison was that of Miller and Miles (1970) who showed that there were minimal differences between the two approaches in terms of species recovery. In both techniques the vegetation is subject to severe disturbance and the recovery of *Calluna* depends on the ability to resprout, which declined with age. In most cases the choice will

depend on ease of application within any management unit and management preferences. Most moorland managers will tend to use prescribed burning because it is cheaper, requires less equipment and for the most part can be applied in almost any situation. Cutting is more expensive, requires expensive cutters which require considerable maintenance when used in moorland situations, and there are some places where cutters cannot be operated easily (stony ground and very steep slopes). There is also potential damage to the peat caused by the cutting operations involving heavy machinery. The other potential benefit to the use of cutting is that it is not constrained to the same extent by weather and legislation, i.e. not constrained by season, although good practice should dictate that cutting should not be done in the bird nesting season. There is also the issue over the cuttings; if they are left *in situ* then the litter can either blow away in very dry conditions or produce a thick, wet mulch which will smother the developing vegetation at least for a time. Where it is left the cut vegetation can act as both a carbon store and a seed source (Ghorbani *et al.*, 2006; Marrs *et al.*, 2007). If the cuttings are removed then cutting becomes an even more costly operation and there is the scope for even further peat damage. However, it is possible that the cut *Calluna* brash can be used or sold either as a seed source for moorland restoration or for use in gulley blocking; which recoups some of the cost.

7.2. Summary of the results from this study

The topic of this thesis is that of prescribed burning with the Peak District National Park. Prescribed burning is a management tool that is used routinely to manage moorland vegetation for grouse and sheep production (Gimingham, 1972). The aim is to remove the above-ground foliage and allow the *Calluna* to resprout from the burned stems. Normally such prescribed burning is done on a rotational matrix, and the aim is to provide a continual supply of moorland vegetation in different stages. The use of prescribed burning for wildfire prevention in other parts of the world is commonplace (Pitkänen *et al.,* 1999). However, in this area there is very little evidence on which to guide management policies and accordingly this study aim to provide information to help develop future policy that is evidence-based (Pullin & Knight, 2009). Specifically, this thesis attempted to answer the following questions:

(a) How degraded are the moorlands in the Peak District, and does prescribed burning affect species density and restoration potential? (Chapter 2)

(b) What are the environmental factors that influence the response of the plant communities, and how do the constituent species respond after prescribed fire? (Chapter 3)

(c) Does prescribed burning affect soil chemical properties? (Chapter 4)

(d) What factors affecting biomass reduction in prescribed fires on upland moorland? (Chapter 5)

(e) What changes in above-ground biomass, carbon and nitrogen occur during prescribed burning? (Chapter 6)

7.2.1. How degraded are the moorlands in the Peak District, and does prescribed burning affect species density and restoration potential? (Chapter 2)

British upland moors are high-priority habitats for conservation. However, the moorlands in the Peak District are particularly depauperate in terms of species composition as they have been severely affected by atmospheric pollution and overgrazing (Tallis, 1972; Anderson & Yalden, 1991). There has also been some policy debate on their future management, in terms of the type of habitat to be conserved and the role of prescribed burning, a commonly used moorland management practice. Chapter 2 addressed the restoration potential of degraded moorland within the Peak District area and assessed the impact of prescribed burning. The study was carried out on five replicate moors in the Peak District National Park, using a rigorous sampling protocol. The vegetation was demonstrated to be severely depauperate relative to the species that might be expected in pristine vegetation reference vegetation, which was derived from the National Vegetation Classification (Rodwell, 1991, 1992).

The NVC is a phyto-sociological framework based on a sample of ca. 31,000 quadrats derived from data obtained from previous studies and a new sampling of high-quality semi-natural vegetation. The NVC classes were derived from a series of multivariate classification analyses to produce "nodal types" of plant communities which cover the range of variation in plant communities found in the UK in the latter half of the twentieth century. This framework has been adopted by the conservation agencies in the UK as the standard for plant community description. As such it is a useful tool, but it is important to remember that the NVC classes are points in a multivariate sea of variation and thus it is only a useful descriptive tool. In this study, the NVC system was used to provide nodal types against which the state of the moor could be judged and specifically which species might be missing, given the depauperate nature of the study moorlands.

The depauperate nature of the moorlands was further evidenced by the fact that the seed bank was also very poor in species diversity. Most species that might be expected to occur in NVC type communities were not in the seed bank, and therefore, propagules must be added to reinstate them. Prescribed burning did impact on the species richness of the moorlands, with an increase immediately after burning and a subsequent decline with time. The hypothesis that a "no-burn" policy would enhance species richness was, therefore, rejected – at least over the 50-year period studied here. Prescribed burning on a rotation less than 20 years, or to maintain a vegetation height of less than 50 cm, was recommended to maintain species richness.

All of the sites studies here have been burned for a considerable period so the studies made here represent a study of one burn cycle imposed on a longer-term series of fires. The study of one burning cycle can of course provide no information on the effects of repeated burning, because it is possible that there will be additive effect which will consistently reduce species diversity. However, the results presented here suggest that prescribed burning can at the very least maintain a greater species diversity than if no burning is applied, and where the prescribed burning in not applied there will be a loss of species.

7.2.2. What are the environmental factors that influence the response of the plant communities, and how do the constituent species respond after prescribed fire? (Chapter 3)

In this chapter the role of prescribed burning for moorland management on plant community composition and its component species was assessed at the regional scale of the Peak District National Park, where the moorland vegetation is severely degraded. Species cover was assessed on five replicated moors with respect to elapsed time since prescribed burning. In each of three years a stratified random selection method was used to choose burn patches covering a range of ages, and quadrats for sampling within those patches. Canonical Correspondence Analysis was used to relate species composition to significant environmental variables and variation partition used to assess their relative contribution. Response curves were produced for the major species with respect to elapsed time since burning and vegetation height.

The species ordination produced two gradients, (a) a continuum from a relatively lichen-rich vegetation to a graminoid-dominated one, and (b) a post-fire growth response of the dominant species, *Calluna*. The five moors covered different sized areas of the CCA plot reflecting their species pool and elevation. The environmental variables explained 13.2% of the variation. Whilst the amount of explained variation is low, it is similar to other studies (Corney *et al.*, 2006). The fraction of the variation that remains unexplained represents a combination of (a) effects of unmeasured environmental variables and (b) stochastic effects. Species composition was more highly correlated with vegetation height than elapsed time since burning. *Calluna* was the only species to show an increasing response after burning; all others either decreased or showed a unimodal\skewed response. Most species with were restricted to vegetation less than 40 cm tall and 22 years since burning. Exceptions *Empetrum nigrum*=47 cm in height and *Eriophorum vaginatum*=37 years after burning.

Therefore, this study suggested that prescribed burning maintained the cover of *Calluna* and appeared to maintain species diversity within a post-burn succession. There was no evidence of colonization either by late-successional species such as *Betula* spp. or bog-forming species such as *Sphagnum* spp. Therefore, it could be argued that prescribed burning is merely maintaining an early-successional dry heath community, however, as the results clearly indicate as the vegetation ages and increases in height, most other species disappear. This evidence presented indicates that prescribed burning is essential to maintain moorland vegetation in its current form, and if it is prevented, the vegetation is likely to become dominated by *Calluna*. Therefore, the recommendation from this work suggests that rotational prescribed burning be continued; the vegetation should be burned when it is between 20-40 cm in height or before 22 years in age. Vegetation height is probably the better yardstick for practical implementation as it is very easy to measure in the field.

All of the evidence presented (Chapters 2,3) suggests that prescribed burning maintains greater plant species diversity than the long-term unburned plots, which become almost a monoculture of *Calluna* with a very deep litter layer. There was no evidence of any species invading these old plots, and whilst it is possible that this would occur in time, there is no evidence of any improvement over a 50-year period. It is unlikely that new species will invade easily given the depth of litter, almost certainly gaps will need to be created to facilitate species regenerating form seed (Miles, 1979). The reduction in the water table of the moorland will almost certainly assist the colonization by later-successional species typical of the drier moorlands but hinder the invasion by bog-forming species.

7.2.3. Does prescribed burning affect soil chemical properties? (Chapter 4)

It is clear that prescribe burning could impact on the soil chemical properties of the peat underlying the moorland vegetation. In this chapter, therefore, the effects of prescribed burning of moorland on soil chemical properties were assessed. Three hypotheses were tested using a chronosequence study replicated on three separate moors, the hypotheses tested were: (1) Does burning have different effects on soils on different moors, either as result of geographical location or management regime? (2) Is there a change in soil chemical properties with time after burning? (3) Is there an interaction with different temporal effects on the different moors? The sampling was done in a structured way within three moorlands. Within-patch variation was relatively low for all soil variables tested.

First, for the majority of soil chemical properties examined, the only significant differences detected were between the replicated moorlands. Second, the impact of prescribed burning and subsequent vegetation recovery had little effect on most of the soil chemical properties measured. Third, some chemical properties did show a temporal relationship with the burn-recovery chronosequence cycle, and two types of response were found, an additive response for the soils' C:N ratios among moorlands, and an interaction with the three moors for four soils properties (available Ca, P, total P, K). The most interesting, and indeed worrying result from an ecological viewpoint was the impact on C:N ratio through time. The C:N ratio was significantly different on the three moors (projected constants: 30.2, 35.1, 32.7 for Bamford, Broomhead and Howden respectively) but declined at the same rate on all three moors (-0.06232 yr^{-1}). This result is counter-intuitive because it would be expected that in the older stands, biomass and litter would accumulate (Chapman, 1967). In these ecosystems the surface soil is highly organic mor humus, classified as a peat (cf. descriptions of similar soils by Heal & Smith, 1978). Hence, the expectation would be that decomposition would be slow and that organic matter would transfer from the litter into the soil. The C:N ratio is a key variable influencing decomposition (Swift *et al.*, 1979; Gundersen *et al.*, 1998); where the C:N ratio is low (<12:1) then decomposition is rapid. As the C:N ratio increases, decomposition declines and peat will be increasingly formed. Here, the oldest stands had the lowest C:N ratio, and whilst they are still well above the threshold limiting decomposition, they are lower than the peats immediately after burning. This may of course reflect an impact of charcoal in the immediate post-burn period.

7.2.4. What factors affect biomass reduction in prescribed fires on upland moorland? (Chapter 5)

Traditionally, prescribed burning aims to maximize *Calluna* regeneration, and hence should ideally remove all above ground vegetation, including moss (Davies *et al.*, 2010), whilst minimizing damage to roots and soil. In the UK, prescribed burns are traditionally lit using a naked flame from a wick or "fire kettle". However, this approach only works well with relatively dry vegetation and is thus limited by weather. The number of days that burning can be carried out is, therefore, limited, and even when burning is possible, burns can only be started late in the day. More recently, a "pressurized-fuel-assisted", approach has been developed where pressurized diesel or gas is sprayed on to the ignition site before lighting. This technique was developed to achieve biomass removal without damage to underlying peat and leaving the bryophyte layer charred, but otherwise intact. The PFA burning technique allows prescribed fires to be lit during wetter conditions, and hence increases the number of days within the burning season when burning can be implemented. Regeneration occurs from resprouting *Calluna* within the remaining bryophyte layer within the first summer. Where the bryophyte layer is wholly removed regeneration tends to be slower (G Eyre, landowner, personal communication). This part of the study considered the relationship between fire severity (assessed through the amount of biomass removed) and (a) fire characteristics, and (b) environmental variables in prescribed burns.

Temperature profiles using thermocouples and pre- and post-burn biomass were recorded in 17 prescribed fires implemented as part of the normal burning regime on Howden Moor, Peak District, UK. The main result was that burn residence time was the most important factor influencing total biomass reduction, with shorter burns removing more biomass. Maximum fire temperature had a negative effect on the reduction of *Calluna* biomass. These results are contradictory to popular understanding of fire behaviour and as such have significant management implications which are discussed. However, the results suggest that the burns with the highest temperatures were flash fires that sped rapidly through the canopy, whereas the burns with the lower residence times were smouldering fires that probably converted more biomass to black carbon (charcoal). This hypothesis needs to be verified by further study.

The ultimate aim of this work was to try and develop a simple series of rules that would allow PFA – burning to be carried out under optimal conditions. Unfortunately, the data are inconclusive, expect insofar as the most efficient for management purposes are the slow smouldering fires.

7.2.5. What changes in above-ground biomass, carbon and nitrogen occur during prescribed burning? (Chapter 6)

One of the main results here suggested that these moorlands do not appear to be following the "traditional *Calluna* life-cycle model" and that regeneration by layering may be an important process (McDonald *et al.*, 1995). This is fundamental to our understanding of long-term change and unfortunately there are no sufficiently old stands available. The use of prescribed burning by definition implies that some biomass, carbon and nitrogen will be lost during burning and this will be accumulated again as the ecosystem recovers. This chapter started to quantify losses and gains.

The study of prescribed burns showed that the loss of biomass during prescribed burning was very variable and this almost certainly reflected a range of factors including meteorological conditions before and during the fire, the fuel load and the its moisture content, the wind speed and the skill of the burning operators (Chapter 5). The PFA-burning approach preferred in the Peak District

(Chapters 5, 6) is designed to minimize biomass loss and maintain as much *Calluna* stick and bryophytes as possible. It was demonstrated that at least in some burns this was very successful.

As the vegetation recovered after burning there was accumulation of above-ground biomass and in these moorlands the oldest stands had much greater biomass values than those reported in the literature (albeit these literature values were for younger sites than the oldest one reported here). In the younger phases of the Peak District vegetation recovery values were similar to literature values this suggests that there is continued accumulation with age. Here the biomass of the oldest stands was more or less confined to the *Calluna* and litter pools.

It is accepted that prescribed burning will release carbon to the environment and here a first approximation has been made to assess the amount and variability of the amounts released. Clearly, a balance needs to be struck between the amounts lost in continued rotational burns relative to the amounts that could be lost in wildfire on older stands where the fuel load may be excessive. Further research on this topic is ongoing, see below.

7.3. Future studies

Whilst the work presented here has provided much improved evidence base for developing moorland management policies there are many areas where further research is needed.

(a) **Burning season, timing of rotations and size of burn.** The burning is restricted by legislation and if climate warms then there may be future restrictions on the length of the season. However, there is good argument in some years for burning in the autumn rather than spring and it may be sensible to allow burning earlier in the autumn into September if the summer is not too dry. The length of rotation has not received much attention, and in the study only very broad-brush limits could be derived (the maximum interval between burns should be less than 20 years). Similarly, the size of burn patch is usually a matter for individual managers, which is based on past experience and safety considerations. There is very little evidence to suggest the optimal size of prescribed burn for conservation benefit.

(b) **Burning within peat landscape dissected with gulleys.** One local problem on the Peak District moor is the difficulty in burning within a dissected peat gulley system. The local conservation agencies have imposed a 5 m buffer around the drainage gulleys and this makes burning around them difficult from a practical viewpoint. Further studies on the impact of prescribed burning around gulleys are needed and specifically linking these impacts to water quality in the drainage water.

(c) **Impact of prescribed burning on ecological processes.** In this thesis a start was made on assessing the impact of prescribed burning on two ecological processes (soil chemistry and seed banks). However, this work was done using a chronosequence approach which is fairly crude one for measuring such processes, and does not take account for any immediate damage during the fire. Future detailed studies immediately before , and after the application of a prescribed burn on a range of processes would be desirable, specifically on (a) soil chemical properties and this to include soil heating, drying and re-wetting, ash production and leaching, (b) heat damage to seed and spore banks, and (c) invertebrate populations.

(d) **Role of prescribed burning in heather beetle management.** Many types of moorland in the United Kingdom are currently under attack from the Heather Beetle (*Lochmaeae suturalis* (Thomson, 1866) (Syrett *et al.* 2000; Rosenburgh & Marrs, 2010). Prescribed sanitation burns have been suggested as one possible way of containing this problems and preventing new outbreaks but these techniques have not been tested experimentally.

(e) **Effects of weather and other factors that influence fire behaviour.** In this thesis data were collected on fire severity (biomass reduction) and related to both temperatures achieved during the fire and meteorological conditions. Unfortunately, temperature data were only available for 17 prescribed burns and meteorological data are extremely difficult to collect in the climatic conditions of the Peak District moors. Hence fewer data were available than expected. This was a pity but the methods adopted to carry out this work showed promise and hopefully can be expanded to other studiers of fire behaviour in the future.

(f) **Development of a whole moor model predicting impacts of prescribed burning within the moorland landscape.** Ultimately from a practitioner point of view there is a need to be able to assess the optimal burning frequency and pattern within a given estate, to help the estate managers develop prescribed moorland management plans. Moorland management plans are a required feature of compliance with the burning legislation (Defra, 2006). One way to do this would be to produce a GIS-based model that incorporates the changing biomass accumulation (fuel load) during the prescribed burning cycle. This approach would allow managers to assess their overall moorland fuel load and its spatial distribution, and hence determine the optimal prescribed burning regime to minimize damage from wildfire. This model is being developed within the BiodivERsA FIREMAN project, and the work presented here will be incorporated into the model.

(g) **Development of a whole moor model predicting impacts of prescribed burning within the moorland landscape.** Whilst carrying out the work for this research it was apparent that there is a great deal of mistrust between many stakeholders, and much of this is as a result of lack of information. It is apparent that many policies are derived and implemented without regard for local knowledge. On the other hand many of the visitors to the Peak District do not understand the need for prescribed burning or indeed the difference between prescribed burning and wildfire. Much more effort needs to be put into assessing how the different stakeholders view prescribed burning within the context of moorland management and to get those different interest groups to communicate more effectively. Thereafter, there is scope for greater interaction between stakeholder groups so that there is good knowledge exchange. There is also a need for education at all levels to ensure that those visitors and indeed local residents understand the importance of the moorlands within an international, national and local context and the need for appropriate management. Again, some aspects of this work us being carried out under the BiodivERsA FIREMAN project.

(h) **Data provision** This project has generated large datasets that provide a baseline for species composition and abundance, seed banks, soil chemistry and biomass for five moorlands in the Peaks District. These datasets are available for modelling purposes and against which future studies can be developed. All of these data are available in electronic form.

7.4. Concluding remarks

The role of prescribed burning is controversial. It is carried out mainly for the management of sheep and red grouse although it has been shown here that prescribed burning can assist in the maintenance of plant species diversity and could provide a mechanism for the control of wildfire. Given future climate change the control of wildfire is likely to be an important factor from a socio-economic viewpoint in the future. From an ecosystem services point of view (carbon sequestration, water quality) the damaging impacts of fire can be viewed as "little and often" (prescribed burning) or "one-off, large and severe" (wildfire). The relative balance in these two strategies applied to any given piece of land will need to be determined by the stakeholders and should reflect societal requirements.

To summarize, from the work presented in this thesis I believe that prescribed burning offers a safe, effective management tool to assist in moor management, specifically to maintain an enhanced flora diversity and a reduced biomass and hence fuel load. When practiced on a rotational matrix the reduction in overall biomass will provide at least some safeguards when a wildfire occurs. The managed moorland will provide:

(a) an overall reduction in fuel which should produce less intense fires and will allow the fires to be easier to bring under control – a '4-hour' rather than '3-day' fire.

(b) A reduced damage to the ecosystem so that there is rapid vegetation recovery and very limited loss of POC and DOC.

The risks of wildfire are likely to increase under current scenarios of global of climate change occur with hotter, drier summers. Thus, the risk of wildfire will inevitably increase in moorland areas where there is a large visitor pressure such as the Peak District National Park. Prescribe burning will, therefore, remain a very important method of moorland management for some time to come.

Chapter 8

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