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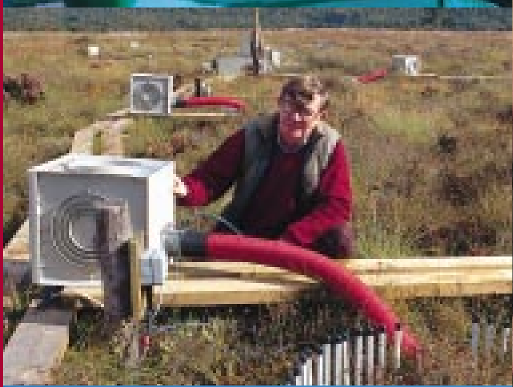
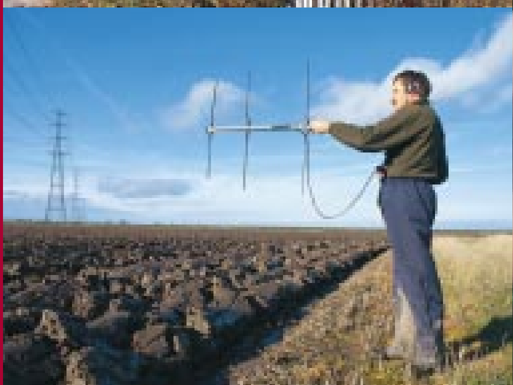


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Review of the impacts of heather and grassland burning in the uplands on soils, hydrology and biodiversity

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Summary

The aim of this study has been to summarise scientific information on the impacts of heather and grassland burning in the English uplands on biodiversity, soils and hydrology. Upland habitats are extensive in England and include a variety of habitats and associated species of conservation concern. This review has therefore given special attention to impacts on such habitats and their associated species.

A detailed literature search and review has been carried out, supplemented with consultations with scientific organisations that have conducted research into upland burning.

Most of the research on managed burning in Britain has been carried out on Scottish grouse moors, in particular studies on moorland burning histories, fire characteristics and impacts on *Calluna vulgaris* (heather) and heathland communities. Many of these research studies date back to the 1960's and 70's. Relatively few scientific studies have been carried out of the impacts of managed fires in England, especially on habitats other than dwarf shrub heaths.

From current research evidence it is apparent that fire has long been a natural component of heathland and other upland ecosystems in Britain. As a result fire-influenced plant communities have developed, which are dominated by those species of plant that have adaptations that enable them to survive fires or rapidly regenerate afterwards. Humans have also used fires since Mesolithic times to clear woody vegetation and manage upland habitats for livestock, and this has been instrumental in the expansion of heathlands, mires and other open upland habitats. Since the early 19th century many heather moorlands have also been burnt for grouse management.

Burning remains a common practice in much of upland England for agricultural and grouse management purposes. The principal aims of burning are to prevent establishment of woody species, to reduce litter and release nutrients, which stimulates earlier growth and temporarily increases the accessibility, palatability, and nutrient content of forage for grouse and livestock. Agricultural burns are usually uncontrolled and therefore tend to cover very large areas in one burn, typically from tens to hundreds of hectares. Grassland burning rotations are often very short, with fires typically returning every couple of years or annually in some areas.

Burning on grouse moors is usually more planned and controlled, and carried out by burning strips of only about 30 m width and typically about 0.5 ha in total area, but larger fires are not uncommon. The normal aim on heather moorland is to burn when most *Calluna* reaches a height of 20 – 30 cm. This should typically result in a fire return period of about 10 – 15 years, in which case 1/10th to 1/15th of the land would be burnt on average each year.

Data on the actual fire frequencies achieved on grouse moors in England are generally lacking, although it appears that there has been a decline in the practice of rotational burning in recent decades on the North York Moors. This situation may not, however, be typical as there is widespread concern in English Nature, based on observations by field staff, that burning management has intensified on some grouse moors, including the frequency of burning rotations.

Burning is legally controlled in England and Wales by the Heather and Grass etc (Burning) Regulations and can only be carried out between the 1st October and 15th April in the uplands. In practice most burning is carried out in the spring due to the requirements for suitable weather conditions.

It is evident from this, and other reviews that, in appropriate areas and circumstances, carefully managed burning can play an important role in the maintenance of some open semi-natural upland habitats in England. Carefully controlled prescribed management burns in appropriate habitats and conditions can arrest succession processes and maintain habitats of high conservation importance such as dry dwarf shrub heathlands. Fires may also help to maintain low nutrient conditions, which is an important characteristic of many upland habitats of high conservation value. And this process may be particularly important under current circumstances where atmospheric pollution results in nutrient inputs from rainfall that are higher than former natural levels. Burning of small patchworks of *Calluna* can also increase vegetation structural diversity and species-richness in plants, invertebrates and birds of *Calluna* dominated heathland habitats (although probably not on peatlands as these habitats have higher inherent levels of diversity). Regular burning also reduces fuel loads and thus to some extent the risks of large and very hot wildfires.

However, the impacts of fires on soils, hydrology and biodiversity are complex and vary according to a number of interrelated factors. Characteristics of the fires are especially important, such as their frequency, temperature, ground surface intensity, residency time and size. These characteristics in turn depend on a range of factors including: fuel type and structure, width of fire, slope, wind and moisture levels in the vegetation and soil, and burning method. Impacts also depend on soil and habitat conditions at the time of burning (which partly reflect the cumulative impacts of burning), season, weather conditions and interactions with grazing and other management practices.

Fires can have significant detrimental impacts, including:

- Ignition, combustion and loss of peat and humus layers by hot fires in dry conditions.
- Increased rates of run-off and erosion, particularly after hot fires and where large or old stands of *Calluna* are burnt, and on steep slopes.
- Reduction in peat accumulation, even under well controlled prescribed burns, and potentially emission of carbon dioxide and other greenhouse gases from carbon stores in peat if these ignite or dry out as a result of hot burns.
- Reduction of structural and species diversity and vegetation composition changes if carried out too frequently or over large areas. In particular, frequent burning dehabilitates *Calluna* and this has probably contributed significantly (together with high grazing pressures) to the loss of *Calluna* cover and replacement by grasses such as *Molinia caerula* (purple moor-grass) over much of the uplands. Burning also exacerbate severely overgrazed habitats if grazing animals are not controlled.
- Post-fire establishment of invasive species such as *Pteridium aquilinum* (bracken), for example where old *Calluna* stands are burnt.
- Destruction and long-term exclusion of fire sensitive and slow colonising species.
- Removal of cover for ground-nesting wildlife and destruction of birds nests and clutches during spring burning periods.

Furthermore, wildfires sometimes develop from poorly set prescribed management fires (*eg* in strong winds or where there is inadequate labour). These can be extremely severe, especially when they occur during dry weather (which is the most likely time). They can extend over very large areas and may also result in combustion of peat leading to soil degradation, severe erosion and long-term ecological and hydrological impacts.

Burning impacts vary considerably according to local circumstances. It is therefore difficult to provide generic recommendations for burning practices for soil and hydrological protection and nature conservation purposes. However, key general recommendations are to:

- Produce burning plans for all areas subject to burning management (for whatever purpose). These should take particular account of impacts on soils and hydrology and the needs of Notified Features of interest within SSSIs and other statutory designated sites.
- Take special efforts to minimise the risk of wildfires, particularly in areas of high biodiversity importance. As many wildfires develop from management fires, special attention should be given to identifying and taking the precautions that are necessary to avoid loss of control; this should form a key part of all burning plans.
- Avoid burning peatland habitats, including wet heath and blanket bog, (unless there is a clear overall conservation benefit for doing so, such as the prevention of <u>severe</u> wildfires which may ignite peat layers <u>in high risk areas</u>) woodlands and scrub, bracken stands, montane habitats, screes and other rocky habitats and flushes. Areas that are used as traditional nest sites by protected species should also not be burnt whatever the habitat.
- Generally avoid burning old stands of *Calluna* as these have low regeneration abilities and are prone to invasion by undesired species once burnt. They are also likely to produce hot fires which create further problems. Such stands should not be brought back into burning management unless there is a clear conservation reason which outweighs the potential risks.
- A general aim for heather moorland management should be to encourage structural and species diversity between and within stands, and the presence of all growth phases of *Calluna*. This should be achieved by a twin track approach, where most *Calluna* dominated stands are carefully burnt in small patches and at a moderate frequency (*eg* when 20-30 cm high and in the building phase) and other selected areas are <u>permanently</u> set-aside from burning. Old *Calluna* should not be provided by simply lengthening rotations.
- Consideration should be given to permanently taking some large areas out of burning management, to allow ‘natural’ succession processes to take place thereby increasing scrub and woodland cover in the uplands and habitat diversity in the long-term. However, the appropriateness, scale and priority for this will depend on wider considerations and debates on social and environmental goals for the uplands.
- Special efforts should be made to reduce the size and frequency of agricultural burns, such as typically carried out on *Molinia* dominated grasslands.
- Burn temperatures on dry heathlands and particularly bogs and wet heaths (if they must be burnt) should be no more than that required to remove most woody vegetation. On grasslands burns should be hot enough to remove the ground litter, but not hot enough to damage the regeneration abilities of any *Calluna* or other shrubs that are present.
- Urgent consideration should be given to shortening the spring burning season to reduce the risks of severe and uncontrolled fires developing in dry conditions and potential nest losses of upland breeding birds. Consideration should also be given, after further research or trials, to extending the autumn burning season into September.

This review also concludes that further research is required on many aspects of burning management in the English uplands. Key research needs include better descriptions of burning practices in each upland area and habitat type, examination of the advantages and disadvantages of extending the burning season into September and shortening the spring burning period, assessments of the impacts of fire on nutrient cycling and budgets, run-off and erosion rates (and in turn their impacts on water courses), bird nesting and the biodiversity of bogs and acid grasslands.

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Contents

1. Introduction

1.1 Aims and scope of the review

The overall aim of this review is to summarise currently available scientific information on the impacts of heather and grassland burning in the English uplands on biodiversity, soils and hydrology, to inform decisions on policy, management and research requirements. Specifically, the objectives of the project are to:

- Conduct a comprehensive search of all relevant literature (and other primary and secondary sources) on the impacts of heather and grass burning practices on upland soils, hydrology, vegetation communities, individual species of higher and lower plants, and all groups of fauna.
- Produce a report summarising and interpreting information resulting from the literature search, which will include information on sources of information and recommendations on priorities for further research.
- Produce a bibliography of scientific information on upland burning.

For the purpose of this review, upland is defined as unenclosed land above 250 m altitude and within Less Favoured Area (LFA) boundaries. The review considers the impacts of burning on the following moorland habitats and associated species of flora and fauna:

- Dwarf-shrub heaths (wet, dry and montane).
- Blanket and other mires, including flushes.
- Scrub, including juniper *Juniperus communis* and western gorse *Ulex galii*.
- Grassland (acid, calcareous and montane).
- Bracken *Pteridium aquilinum*.
- Scree and rock habitat.

Indirect impacts on other habitats, such as effects of changes in water flow and increased soil erosion on water bodies, are also be considered where appropriate.

All scientific information on potential impacts on upland soils, hydrology and biodiversity has been examined. However, particular attention has been given to assessing and reporting on the effects of the frequency, intensity and timing of burning, burning methods, size of burns, aspect, altitude and other geographical effects on burning impacts.

A detailed examination of impacts from wildfires resulting from accidents and arson is not provided here. This is because the aim of this review is to inform policies on legal burning management practices. However, studies of wildfires, their impacts and habitat recovery are examined where they provide information that is relevant to consideration of prescribed burning practices.

1.2 Methods and sources of information

1.2.1 Identification of literature sources and preparation of bibliography

This review has primarily been undertaken as a desk study of existing data and literature sources. The first part of the project was, therefore, the preparation of a comprehensive bibliography on burning impacts in the uplands. This concentrated on identifying the original outputs of scientific studies, rather than secondary reviews, summaries and management guidelines, unless these provide the only traceable source of information on particular issues. Such scientific studies have included empirical observational studies (*eg* comparisons of upland vegetation composition and structure in relation to their management and burning history), experimental studies (*eg* where vegetation changes have been observed before and after experimental burning treatments) and modelling studies. Key sources of background information on, for example, the historic use of fire as a management tool , are also identified and included in this review.

There is an extensive body of literature on general fire ecology in various habitats worldwide, but it has been beyond the scope of this review to examine this thoroughly. Some key reviews of this subject and selected references are, however, referred to where appropriate. Studies conducted outside England, are also included in this review where these have examined the impacts of burning on similar habitats to those found in England. Indeed, much of the relevant research considered here comes from studies carried out on the intensively fire managed grouse moors of north-east Scotland.

This literature review has examined scientific material on all the habitats listed in Section 1.1 where these occur in the uplands. However, some of these habitats have similar counterparts in the lowlands, and studies of these provide some information which is also relevant to burning impacts in the uplands. In particular, there is an extensive literature on lowland heathland, which has many similarities to upland dwarf shrub heaths. This review has therefore attempted to identify and use key scientific studies of lowland habitats where their findings may be applicable to upland situations, particularly where analogous upland studies and data are lacking. But given the potential range of UK and European studies across all lowland heath, scrub, mire, grassland, bracken and rocky habitats, it has not been possible to identify or assess all potentially relevant sources of information from these habitats.

The bibliography has been compiled by firstly identifying relevant references from previous reviews of upland ecology and management that have included analysis of burning impacts and related issues. In particular the bibliography has built on a previous English Nature Research Report, *Literature review of the historical effects of burning and grazing of blanket bog and upland heath wet heath* (Shaw *et al* 1996) and the English Nature *Upland management handbook* (Backshall *et al* 2001). This has been added to by checking the contents of previous issues of the ecology, soil science and hydrology scientific journals that have most frequently included papers on burning in the uplands and related subjects. These journals included Biological Conservation, Journal of Animal Ecology, Journal of Applied Ecology, Journal of Ecology, Journal of Environmental Management and Vegetatio. These were mostly checked via web-based online searchable databases.

Further computer based online searches were also carried out using the following selected bibliographic databases: Incarta, Cambridge Scientific Abstracts, Cambridge University Library database, English Nature WildLink and (via the Dialog system) GeoBase and Enviroline.

These online database searches were carried out using selected key words to identify potentially relevant references. The search terms were references containing in the title, keywords or abstract:

- burn or muirburn[1] or fire or swale[2] or swaling

  AND

- upland or moor or heath or bog or mire or *Calluna* or *Erica* or *Molinia* or *Eriophorum* or *Nardus* or *Sphagnum* or grouse or peat or soil or hydrology.

The database derived lists of references containing these combinations of key words were then screened by checking each reference's title and abstract. Standard reference citation details were then extracted from each literature source that was considered likely to hold relevant scientific information on the impacts of burning in the uplands. These were then entered into an Adept Scientific EndNote 6 library database. Appropriate key words describing the content of the publication were also deduced and entered, as well as the abstract for selected key references. This electronic database is, therefore, searchable by subject area, as defined by key words. It is available as a supplement to this report on request to English Nature.

1.2.2 Bibliography checking and consultations

The draft bibliography was then circulated to key organisations and research institutes involved in upland research and management (see Appendix 1). These were identified from recent papers in the draft bibliography and consultations with English Nature and DEFRA staff. Each consultee was invited to check that all their relevant research outputs are identified and listed correctly, and to provide information on missing references if they desired.

This review has also collected basic information describing some current research studies on upland burning impacts and closely interrelated issues. This was collated by circulation of a simple questionnaire to the key research consultees listed above. Completed questionnaires are presented in Appendix 2. However, because not all consultees responded, this should not be regarded as a comprehensive list of current research on the subject.

In addition to the desk study and correspondence, two brief site familiarisation and consultation visits were made. Firstly to an area of heather moorland primarily managed for grouse in Swaledale, Yorkshire, where consultations were held with Colin Newlands and Peter Welsh (English Nature) and Des Coates (Head Game Keeper of the Grinton Moor Estate). Secondly to a range of heather moorland, south-west heath and upland acid grassland habitats in Dartmoor National Park, where consultations were held with Simon Bates and Albert Knott (English Nature), Maurice Retallick (local farmer and Council Member of Dartmoor National Park Authority) and Suzanne Goodfellow (Dartmoor National Park Authority). In addition a consultation meeting was held with Peter Stevens and Dave Glaves of Exeter DEFRA Rural Development Service (RDS).

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[1] The term used for the management of moorland vegetation by burning in Scotland

[2] The term used for the management of vegetation by fire in the south-west

1.2.3 Terminology

A glossary of frequently used terms is provided in Appendix 3. Fire terms are used in this report according to the meanings given in *Prescribed burning on Moorland: Supplement to the Muirburn Code: A Guide to Best Practice* (Anonymous, 2001).

Plant species are only referred to in this report using their scientific names, to avoid confusion over possible alternative English names. Plant taxonomy and nomenclature follows that used in Stace (1997). A list of National Vegetation Classification (NVC) communities (Rodwell 1991a, b, 1992, 1995, 2000) referred to in this report is provided in Appendix 4. Only scientific names are used for invertebrates. Other animals are referred to using their English name and are listed together with their scientific name in Appendix 5.

1.3 Upland habitats and their biodiversity importance

1.3.1 Introduction

The uplands cover almost a third of Britain and include the largest areas of semi-natural habitat, mainly comprising a range of sub-montane anthropogenic grassland and dwarf shrub habitats (Birks 1988). Remaining natural climatic 'climax' communities are predominantly extensive ombrogenous blanket bog or montane heath (the latter being very restricted in England). Climate is the major factor affecting the types of habitat present, which in turn is principally influenced by altitude, latitude and increasing oceanicity from west to east. Local variations in habitat then reflect differences in geology, topography, past human impacts and current uses and management.

An important affect of human activities is that the habitats within the upper altitudinal range of tree growth have been profoundly modified. The sub-alpine – alpine zone of *Salix*, low *Juniperus* and dwarf *Betula* scrub, which would typically occur under natural conditions, has been largely eradicated (Birks 1988). Patches of *Betula* and *Juniperus* scrub occur but these do not form climax communities and *Salix* scrub only exists on cliff ledges. As a result the natural tree-line can be seen only as a few fragments and the upper limits of remaining woodland are mostly well below their potential altitudinal range. Thus, today a typical upland vegetation zone in England consists of meadows, enclosed pastures and woodlands in valley bottoms and on lower slopes, up to sub-alpine communities of dwarf shrub heaths, grasslands and blanket bogs (with patches of gorse or other scrub in some locations) and alpine communities of grassland or rocky moss- or lichen-heath above the climatic tree-line at approximately 650 m (though this varies by +/- 100 m depending on exposure).

Although generally anthropogenic, species poor and relatively widespread in the UK, several upland habitats are internationally scarce and therefore a high conservation priority (Birks 1988, Usher and Thompson 1993, Ratcliffe and Thompson 1995, Thompson *et al* 1995a, Thompson *et al* 1995b, Backshall *et al* 2001). An important aim of this review is, therefore, to consider the impacts of burning on these particular habitats and their associated species of conservation concern. There are currently a number of lists and criteria for identifying such habitats and species, but to assist with statutory requirements this review has given particular attention to assessing impacts on terrestrial habitats listed in Annex I of the Habitats Directive (*EU Council Directive 92/43/EEC on the conservation of natural habitats of wild fauna and flora*). Consideration is also given to impacts on UK Biodiversity Action Plan (UK BAP) Priority Habitats that occur in the uplands of England.

Consideration of species in this review focuses on impacts on overall assemblages and diversity within taxa groups, species listed on Annex I of the Wild Birds Directive (*Council Directive 79/409/EEC on the conservation of wild birds*) and Annex II of the Habitats

Directive, and UK BAP Priority Species (although there are few). It should be remembered that many other species will also be of some degree of conservation importance (at national, regional and local scales) and it is not possible to give a comprehensive assessment of all burning impacts on these in this review.

English upland habitats that are subject to regular intentional burning management are briefly described below, under UK BAP Broad Habitat headings, and their associated Habitats Directive Annex I and UK BAP Priority habitats are listed in Table 1.1 together with their corresponding NVC types (see Appendix 4 for details). Current conservation targets for the UK BAP Priority habitats listed in Table 1.1 are provided in Appendix 6. Woodlands other than *Juniperus* woods and neutral grasslands (typically upland hay meadows) are not considered here as these are not subject to deliberate burning management.

Table 1.1 Habitats Directive Annex 1 Habitats and UK BAP Priority habitats and their NVC equivalents that occur in the uplands of England and are subject to burning management (based on Brown *et al* 1997, Drewitt and Manley 1997, Jackson 2000, Jackson and McLeod 2002)

UK BAP Broad Habitat (bold) / Habitats Directive Annex 1 type	Annex I code	EU Priority Habitat	UK special responsibility	NVC types in uplands in England (see Appendix 4 for details)	UK BAP Priority habitat[*1]
Broadleaved, mixed and yew woodland / coniferous woodland[*2]					
Juniperus communis formations on heaths or calcareous grasslands	5130			W19, W21	Upland heathland / upland calcareous grassland where components of these habitats
Calcareous grassland					
Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (important orchid sites)	6210			CG2, CG9	Upland calcareous grasslands
Species rich *Nardus* grassland, on siliceous substrates in mountain areas (and submountain areas, in continental Europe)	6230	Yes	Yes	CG10, CG11 (and some species-rich stands of U4 & U5)	Upland calcareous grasslands
Dwarf shrub heath					
Northern Atlantic wet heaths with *Erica Tetralix*	4010		Yes	M15, M16, H4, H16, H21	Upland heathland
European dry heaths	4030		Yes	H4, H9-10, H12, H16, H18, H21	Upland heathland
Fen, marsh and swamp					
Molinia meadows on calcareous, peaty or clayey-silt-laden soils (*Molinion caeruleae*)	6410			M26	
Transition mires and quaking bogs	7140			M5, M8, M9 & others	
Petrifying springs with tufa formation (*Cratoneurion*)	7220	Yes		M37, M38	

UK BAP Broad Habitat (bold) / Habitats Directive Annex 1 type	Annex I code	EU Priority Habitat	UK special responsibility	NVC types in uplands in England (see Appendix 4 for details)	UK BAP Priority habitat[*1]
Alkaline fens	7230			M9, M10, M13	
Alpine pioneer formations of *Caricion bicoloris-atrofuscae*	7240	Yes		M10, M11	
Bogs					
Active raised bogs	7110	Yes	Yes	Includes M1, M2, M18, M19 & others	
Degraded raised bogs still capable of natural regeneration	7120			Includes M3, M15, M16, M20, M25 & others	
Blanket bog (active only)	7130	Yes	Yes	M1, M15, M20, M25 & others	Blanket bog
Montane habitats					
Alpine and boreal heaths	4060		Yes	Alpine heaths: H13, H19 Boreal heaths subalpine forms of: H10, H12, H16, H18, H21	
Siliceous alpine and boreal grassland	6150		Yes	U7 & U10	
Inland rock					
Calaminarian grasslands of the *Violeralia calaminariae*	6130		Yes	In the UK some forms of this vegetation correspond to NVC OV37 but others occur in ecotypes not characterised by the NVC	
Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels	6430			U17	
Siliceous scree of the montane to snow levels (*Androsacetalia alpinae* and *Caleopsietalia ladani*)	8110			U21	
Calcareous and calcshist screes of the montane to alpine	8120			OV38 and other forms with no NVC	

UK BAP Broad Habitat (bold) / Habitats Directive Annex 1 type	Annex I code	EU Priority Habitat	UK special responsibility	NVC types in uplands in England (see Appendix 4 for details)	UK BAP Priority habitat*1
levels (*Thlaspietea rotundifolli*) (also known as Euric scree)				equivalents	
Calcareous rocky slopes with chasmophytic vegetation	8210			OV39, OV40 and other forms with no NVC equivalents	
Siliceous rocky slopes with chasmophytic vegetation	8220			U21 and other forms with no NVC equivalents	
Limestone pavements	8240	Yes	Yes	A wide range of calcicolus NVC types occur	Limestone pavements

Notes. *1 UK BAP Priority Habitats are: habitats for which the UK has international obligations (eg under the Habitats Directive); habitats at risk, such as those with a high rate of decline, especially over the last 20 years, or which are rare; habitats which may be functionally critical (ie areas that are part of a wider ecosystem but provide reproductive or feeding areas for particular species); and habitats which are important for UK BAP Priority Species.

*2 *Juniperus* formations on calcareous grassland are included in the "Broadleaved, mixed and yew woodland" Broad Habitat; whereas juniper formations on heaths are included in the "Coniferous woodland" broad habitat (Jackson 2000).

1.3.2 Broadleaved, mixed and yew woodland / coniferous woodland

This Broad Habitat includes scrub vegetation, (where the woody component tends to be mainly shrubs usually less than 5 m high), which includes *Juniperus* stands (see Table 1.1) and these may be subject to burning as part of heathland and grassland management.

Juniperus occurs on acidic and calcareous substrates and in a variety of vegetation communities, with some stands being associated with typical examples of heath or calcareous grassland vegetation. In some cases they occur as transitional habitats between woodland/scrub and calcareous grassland, heath, acidic grassland, rock outcrops, limestone pavements, scree and cliffs.

However, where *Juniperus* has been long established, a more distinctive scrub or woodland habitat assemblage may occur, in which *Juniperus* is associated with other shrubs, shade-tolerant herbs, grazing-sensitive tall herbs, bryophytes and ferns. On acidic soils (mainly in northern England) an NVC type W19 *Juniperus communis ssp. communis – Oxalis acetosella* woodland may develop. This is typically dominated by *Juniperus*, with *Betula pubescens* and *Sorbus aucuparia* often scattered throughout. The understorey is rich in acidophilous species and there is usually a well-developed layer of pleurocarpous mosses and ferns. On thin chalk soils closed *Juniperus* stands with a rich scrub flora occur, corresponding to NVC type W21d *Crataegus monogyna – Hedera helix* scrub, *Viburnum lantana* sub-community. This form of *Juniperus* scrub is more commonly found in lowland southern England, but is also found further north, at higher altitude on limestone, where *Juniperus* scrub is often associated with limestone pavements, calcareous cliffs and screes.

This Broad Habitat type does not include gorse *Ulex minor* and *Ulex gallii* scrub which is included in the "Dwarf Shrub Heath" Broad Habitat type and montane willow scrub which is included in the "Montane Habitats" Broad Habitat type.

1.3.3 Calcareous grassland

Calcareous grasslands develop on shallow lime-rich soils usually derived from limestone or chalk. Swards tend to be much more species-rich than upland grasslands on acidic substrates, and may contain over 60 species/4m^{-2}. They support a series of widespread grassland plants which are mainly restricted to calcareous soils, but also a high diversity of rare plants. For example, upland and mountain grasslands are rich in arctic-alpines, such as *Gentiana verna* and *Dryas octopetala*.

Scrub is also an important associated feature of many calcareous grasslands. Certain types of calcareous scrub, such as *Juniperus* scrub, and the species-rich *Corylus avellana* scrub of the Derbyshire Dales have a high intrinsic conservation value and are rare. But, in some areas, without appropriate management, scrub encroachment may threaten some grasslands.

As noted in Table 1.1 two Annex 1 Habitats fall under this Broad Habitat category (and associated *Juniperus* scrub) and Upland Calcareous Grasslands have been identified as a UK BAP Priority Habitat. The Priority Habitat includes six NVC communities of which four occur in the uplands of England: forms of *Sesleria albicans* grassland (CG9); *Festuca ovina - Agrostis capillaris* swards (CG10 and CG11); and *Dryas octopetala* communities (CG14). There is an estimated total of approximately 22,000-25,000 ha of Upland Calcareous Grassland in the UK, of which about 10,000 ha is in England. Particularly important areas for the habitat in England include the Craven Pennines in the Yorkshire Dales and upper Teasdale in the northern Pennines (Drewitt and Manley 1997).

Upland Calcareous Grassland is an important habitat for a number of UK BAP Priority Species including the mason bees *Osmia inermis* and *O. parietina*, the ruby-tailed wasp *Chrysura hirsuta*, the snails *Vertigo geyeri* and *V. genesii*, and the plant *Alchemilla minima*.

1.3.4 Acid grassland

Acid grasslands are probably one of the most extensive semi-natural habitats in Britain, occurring on sandstones, acid igneous rocks and superficial deposits such as sands and gravels. Particularly large expanses of acid grassland occur in the uplands.

Although the habitat is typically species-poor a range of communities occur in the UK. Of these six NVC types of acid grassland occur in upland areas in England (Drewitt and Manley 1997): U1 *Festuca ovina-Agrostis capillaris-Rumex acetosella* grassland; U2 *Deschampsia flexuosa* grassland; U3 *Agrostis curtisii* grassland; U4 *Festuca ovina-Agrostis capillaris-Galium saxatile* grassland; U5 *Nardus stricta-Galium saxatile* grassland; and U6 *Juncus squarrosus-Festuca ovina* grassland.

These habitats typically have a limited biodiversity interest and may occur in large uniform expanses, particularly in the uplands and where overgrazing and frequent burning has occurred (see later discussions). However, they can add to the diversity, structure and function of moorland landscapes where they occur as part of a mosaic with other habitats such as heathland and mires. Some more species-rich acid grassland types also occur (such as U4c and U5c) but these are restricted in distribution, occurring for example in the North Pennines (Backshall *et al* 2001).

1.3.5 Bracken

This Broad Habitat type consists of areas dominated by a continuous canopy cover of *Pteridium aquilinum* at the height of the growing season. It is the overwhelming dominant species in the U20 *Pteridium aquilinum-Galium saxatile* community, which grows with species-poor acidic grassland and dwarf shrub communities. *Pteridium* also often forms dense monoculture stands, where a build up of acidic litter eliminates understorey vegetation.

Pteridium stands are commonly viewed as of low conservation importance (Pakeman and Marrs 1992) and an agricultural problem as the species is highly invasive and reduces grazing areas. However, it does provide habitat for some species of conservation importance, including the high brown fritillary butterfly *Argynnis aglaia* and whinchat, and may therefore be of value when part of upland habitat mosaics.

1.3.6 Dwarf shrub heath

This habitat includes 3 listed Annex 1 Habitats that occur in the uplands and is covered in the uplands by the UK BAP Upland Heathland Priority Habitat. The Upland Heathland Habitat Action Plan (HAP) defines the habitat as vegetation characterised by the presence of dwarf shrubs at a cover of at least 25% lying below the alpine or montane zone (at about 600-750 m) and usually above the upper edge of enclosed agricultural land (generally at around 250-400 m, but descending to near sea-level in northern Scotland).

The habitat occurs widely on mineral soils and thin peats (<0.5 m deep) throughout the uplands of the UK. The vegetation varies depending on the influence of climate, altitude, aspect, slope, maritime influences and management practices, including grazing and burning. A wide range of heathland NVC communities occur in English uplands, including *Ulex gallii - Agrostis curtisii* (H4), *Calluna vulgaris - Ulex gallii* (H8), *Calluna vulagris - Vaccium myrtillus* (H12), *Calluna – Erica cinerea* (H10), *Calluna - Vaccinium myrtillus - Sphagnum capillifolium* (H21), *Trichophorum cespitosum – Erica tetralix* (M15) and *Vaccinium myrtillus - Deschampsia flexuosa* (H18).

High quality heaths are generally structurally diverse, containing stands of vegetation with heather at different stages of growth. Upland heath in 'favourable condition' usually includes areas of mature heather and is typically dominated by a range of dwarf shrubs such as *Calluna vulgaris* (hereafter refereed to in this review as *Calluna*), *Vaccinium myrtillus*, *Empetrum nigrum*, *Erica cinerea* and, in the south and west, *Ulex gallii*. In northern areas *Juniperus communis* may also occur above a heath understorey. Wet heath is most commonly found in the wetter north and west and is typicaly dominated by mixtures of *Erica tetralix*, *Trichophorum cespitosum*, and *Molinia caerulea* (hereafter referred to as *Molinia*), over an understorey of mosses often including carpets of *Sphagnum* species.

The habitat is not particularly species rich, but does support characteristic faunal assemblages (Usher and Thompson 1993, Ratcliffe and Thompson 1995) including upland birds, such as red grouse, black grouse (though now rare), merlin and hen harrier. Some forms of heath also have a significant lower plant interest, including assemblages of rare and local mosses and liverworts that are particularly associated with the wetter western heaths. The arthropod fauna can also be highly diverse, particularly at lower altitudes, and includes some nationally rare species (Usher 1992).

Priority Species identified as part of the UK BAP currently include black grouse, a cranefly *Tipula* (*Savtshenkia*) *serrulifera*, the moths *Semiothisa carbonaria*, *Xestia alpicola alpina* and *Xylena exsoleta*, and *Juniperus communis*.

1.3.7 Fen, marsh and swamp

These habitats, which are common in the uplands, are characterised by a variety of vegetation types that are found on minerotrophic (groundwater-fed), permanently, seasonally or periodically waterlogged peat, peaty soils, or mineral soils. They include 5 Annex 1 listed habitats (Table 1.1).

Molinia meadows are found mainly on moist, moderately base-rich, peats and peaty gley soils, often with fluctuating water tables. The upland form is M26 *Molinia caerulea – Crepis paludosa* mire, which in England occurs locally in northern wet grasslands and fens in uplands and upland margins. The vegetation has a distinctive sub-montane character and include species with a northern distribution, such as *Crepis paludosa* and *Trollius europaeus*.

Transition mires and quaking bogs occur in a variety of situations: in flood plain mires, valley bogs, basin mires and the lagg zone of raised bogs, and as regeneration surfaces within mires that have been cut-over for peat or areas of mineral soil influence within blanket bogs. Their floristic composition and general ecological characteristics are transitional between acid bog and alkaline fens and therefore their plant communities normally exhibit intimate mixtures of acidophilous, calciphilous or basophilous species.

Alkaline fens consist of a complex assemblage of vegetation types characteristic of sites where there is tufa and/or peat formation with a high water table and a calcareous base-rich water supply. The core vegetation is short sedge mire of the following NVC types: M9 *Carex rostrata – Calliergon cuspidatum/giganteum* mire; M10 *Carex dioica – Pinguicula vulgaris* mire; and M13 *Schoenus nigricans – Juncus subnodulosus* mire.

There is considerable variation between alkaline fens primarily as a result of their geomorphological situation, but also altitude, with upland fens being rich in northern species. A significant proportion of the habitat surviving in the EU occurs in the UK, where it has declined dramatically in the past century. It is now widely and unevenly scattered with mostly small, isolated fragments of habitat. It is found in lowland situations but important upland sites occur in northern England.

Petrifying springs with tufa formation (*Cratoneurion*) are habitats associated with hard-water springs, where calcium bicarbonate rich groundwater comes to the surface and leaves a hard deposit of calcium carbonate (tufa). These conditions occur most often in areas underlain by limestone or other calcareous rocks. Tufa formation is a relatively rare in the UK and petrifying springs occur as small, scattered flushes, mainly in the uplands of northern England and the Scottish Highlands.

Tufa-forming spring-heads are characterised by the swelling yellow-orange mats of the mosses *Cratoneuron commutatum* and *C. filicinum*. The two main NVC habitat types are M37 *Cratoneuron commutatum – Festuca rubra* springs and M38 *Cratoneuron commutatum – Carex nigra* springs. The former community is widely distributed, while the latter is found only at moderate to high altitudes and has a flora especially rich in rare arctic-alpine species, which is best-developed in upper Teesdale and the Scottish Highlands.

Alpine pioneer formations of the *Caricion bicoloris-atrofuscae* is a type of flush mire that occurs only at high altitude, where the characteristic plant communities are constantly flushed by surface seepage of cold, base-rich water. It is one of the few remaining natural plant communities in the UK and is characterised by the presence of a number of rare specialised species. The habitat is rare and largely restricted to the Scottish Highlands, but there are southern outliers in northern England and north Wales. Given its location the habitat is unlikely to be affected by burning.

The UK BAP Broad Habitat also includes flushes, which are associated with lateral water movement, and springs with localised upwelling of water; these often form important local features on moorland landscapes.

1.3.8 Bogs

Bogs (or ombrotrophic mires) are particularly characteristic of the uplands and form amongst the most extensive areas of semi-natural habitat in the UK. They include a variety of habitat types that usually support peat-forming vegetation and receive mineral nutrients principally from precipitation rather than ground water. If not degraded by drainage the water-table is normally at, or just below, the surface, and they are typically dominated by acidophilous species such as *Sphagnum* spp., *Eriophorum* spp. and *Erica tetralix* The habitat often forms complex mosaics with associated communities, such as flushes, fens and swamps.

Bogs are of considerable conservation importance in the EU and the UK and include two Annex 1 Priority Habitats (Table 1.1). Blanket Bog[3] is also a UK BAP Priority Habitat which includes NVC types M1-3, M15, M17-20 and M25, together with their intermediates. This UK BAP Priority Habitat encompasses the Habitats Directive Annex 1 Priority Habitat 'Active' Blanket Bog, the definition of active being given as “still supporting a significant area of vegetation that is normally peat forming”.

When in good condition blanket bogs support a very wide range of terrestrial and aquatic vertebrates and invertebrates. As with plant species, some of these are widespread and common, some are much more local, and quite a number are of international importance, in terms of their rarity or the densities of their breeding populations. However, none of the Priority Species listed under the UK BAP are principally associated with this habitat type.

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[3] Note that the term blanket 'bog' strictly applies only to the portion of a blanket 'mire' which is exclusively rain-fed, but the HAP regards the terms 'bog' and 'mire' as more or less synonymous.

1.3.9 Montane habitats

Montane habitats include a range of vegetation types that occur exclusively in the montane zone such as prostrate dwarf shrub heath, snow-bed communities, sedge and rush heaths, and moss heaths. However, the distinction between the sub-montane and montane zone is often blurred and the two usually merge through a band of transitional vegetation (Jackson 2000). Exclusively montane habitat types can be recognised by their floristic composition and their typical prostrate vegetation structure.

Two montane habitats are listed on Annex 1 of the Habitats Directive: Alpine and boreal heaths, and siliceous alpine and boreal grassland.

Alpine heaths develop above the natural altitudinal tree-line whilst boreal heaths develop below the tree-line where woodlands have been removed (or in gaps between woodland stands in natural conditions). There are two broad types of alpine and boreal heath, dominated either by *Calluna* on exposed or more sheltered ground at lower altitudes, or, by *Vaccinium myrtillus* and *Empetrum nigrum* at higher altitudes beyond the limit of heather growth or in more sheltered localities where heather growth is suppressed by prolonged snow-lie (Jackson and McLeod 2002). A number of rare arctic-alpine plants occur and exposure or snow-lie, which suppress the growth of dwarf-shrubs, also favours the growth of characteristic lichens and bryophytes. Some forms of the habitat support Atlantic mosses and liverworts that are found with restricted world distributions. Alpine and boreal heaths that are rich in bryophytes, lichens and *Juniperus* are susceptible to disturbance, especially by fire.

Siliceous alpine and boreal grasslands are of particular importance in the UK as they are one of the few remaining predominantly near-natural habitats, and the UK and Ireland support the only examples of this vegetation within the Atlantic biogeographical region (Jackson and McLeod 2002). The habitat is the most extensive type of vegetation in the high mountain zone, ie above an altitude of about 750 m. It comprises a range of grassland types whose composition is influenced by contrasting extremes of exposure and snow-lie, and includes late-lie snow-bed communities dominated by bryophytes and dwarf-herbs.

The most extensive and best-developed examples of montane habitats are found in the Scottish Highlands, but outliers also occur on the higher mountains of England.

1.3.10 Inland rock

A large proportion of the uplands consists of rocky areas, which despite their apparent lack of vegetation, can support important plant and animal communities and species of conservation interest. Indeed, seven types of rocky habitat are listed in Annex 1 of the Habitats Directive (Table 1.1).

These listed habitats include limestone pavement, which is a Habitats Directive Priority Habitat and also a UK BAP Priority Habitat. Limestone pavements are areas of exposed limestone that have been affected by weathering to form a complex structure of massive blocks of worn limestone (clints) interspersed with crevices (grikes). The vegetation is rich in vascular plants, bryophytes and lichens, and varies according to geographical location, altitude, rock type and the presence or absence of grazing animals. Limestone pavement vegetation may also contain unusual combinations of plants, with woodland and wood-edge species well-represented in the sheltered grikes. The clints support plants of rocky habitats or are often unvegetated. In the absence of grazing scrub often develops. They are of geological as well as biological importance.

The UK holds a significant proportion of the EU resource of this habitat, although it is scarce and widely scattered, primarily occurring in England in North Yorkshire and Cumbria, with smaller areas in Lancashire and elsewhere.

Other Annex 1 upland rocky habitats include a range of chasmophytic vegetation (plant communities that colonise the cracks and fissures of rock faces); calaminarian grassland (a grassland type which is found on soils which have high levels of heavy metals, such as lead, chromium and copper, that are toxic to most plant species); and certain types of grazing sensitive tall herb and fern communities, that are confined to inaccessible cliff faces and ledges, steep rocky slopes and boulder fields. Both siliceous scree and calcareous and calcshist screes of the montane to alpine levels are important for their rich fern flora and act as refugia for a number of rare species.

1.4 The protection and condition of upland habitats in England

As a result of their biodiversity, geological and landscape importance a high proportion of the uplands in England have been designated as National Parks, Areas of Outstanding Natural Beauty (AONBs), Sites of Special Scientific Interest (SSSIs) and National Nature Reserves (NNRs), and therefore receive protection under The National Parks and Access to The Countryside Act (1949), The Wildlife and Countryside Act (1981), The Wildlife and Countryside Amendment Act (1985) and the Countryside and Rights of Way Act 2000 (CRoW Act). A substantial number of these areas are also of international importance and have received additional protection as Special Protection Areas (SPAs), under the EU Council Directive 79/409/EEC on the Conservation of Wild Birds (known as the Birds Directive), and Special Areas of Conservation (SACs), under EU Council Directive 92/43/EEC on the Conservation of Natural Habitats and of Wild Fauna and Flora (known as the Habitats Directive).

A large proportion of the uplands also fall within Environmentally Sensitive Areas (ESAs), which form part of the England Rural Development Programme. Within ESAs landowners can obtain grants under 10-year agreements to follow land management prescriptions that aim to protect or enhance wildlife, archaeology and landscape values. There is, however, some evidence that these measures have had limited success in maintaining habitats in favourable condition or enhancing habitats where they have been previously degraded (Reid and Grice 2001, Ecoscope Applied Ecologists *et al* 2003).

In the past there has been substantial losses of many of the upland habitats of high conservation value, primarily through agricultural improvements and afforestation (Thompson *et al* 1995b). However, as a result of their protection and the reduced economic returns from upland forestry and agriculture, losses of many upland habitats have now declined (Haines-Young *et al* 2000). But there remain significant conservation problems in the uplands (Bardgett *et al* 1995, Brown *et al* 2001). According to English Nature the main threat to uplands habitats is overgrazing, primarily by sheep, as livestock numbers have increased in recent years following the introduction of headage support under the EU Common Agricultural Policy (Brown *et al* 2001). Inappropriate burning, especially on blanket bogs, is also considered by English Nature to be a serious problem. This largely stems from a belief that there has been a decline in good burning practices in recent years (Phillips & Watson 1995) despite the availability of heather burning guidelines for some years (MAFF 1994, Anonymous 2001b and previous versions). In some areas heather burning appears to be less frequent than advised in burning guidelines (eg in Scotland according to Hester and Sydes 1992, Phillips and Watson 1995), whilst in others it occurs more frequently and over more extensive areas than recommended (eg on Dartmoor according to Ward 1972). There has also been a number of large uncontrolled fires in recent springs on grouse moors in England. One such fire in spring 2003 covered 250 ha of the Bowland Fells, and burnt out the most important moorland nesting area for breeding hen harriers in England. The fire destroyed two active hen harrier nests and possibly a third nest of a female that appeared to be settling in the area. In fact it appears that some of these spring fires may have been deliberately set to destroy harrier nesting habitat. English Nature state that at least one of the burns in the North Pennines was intentionally targeted at last year's nest site, to prevent the birds settling again in 2003 (English Nature Press Release, 14 April 2003).

Other threats to upland habitats include increased access and recreation, eutrophication and acidic deposition resulting from atmospheric pollution, and ecological changes may also be arising from climate change (Brown *et al* 2001).

These pressures have had a significant impact on the vegetation composition of moorland, with declines in the cover of *Calluna* and other dwarf shrubs and an increase in competitive grasses such as *Molinia* (Welch and Scott 1995) resulting in the replacement of areas of dwarf-shrub heath and bog by acid grassland. In a study of the extent and condition of heather on moorland in England and Wales it was found that 440,000 ha of upland heathlands had less than 25% heather cover (Bardgett *et al* 1995). Within England the most suppressed heather was found to occur in the west Midlands and the south-west. In contrast, heather moorlands in the North York Moors, north-east and north-west England were generally in good condition and remained the dominant vegetation type over large areas.

The condition of SSSIs is now monitored under a Common Standards framework agreed between the UK Statutory Conservation Agencies and the Joint Nature Conservation Committee (JNCC 1998). Under this framework a standard set of conditions including "Favourable Condition" and variations of "Unfavourable Condition" have been defined. Condition assessments are then carried out by the statutory conservation agencies using standardised sets of condition assessment criteria for the habitat or species in question.

A current DEFRA Public Service Agreement (PSA) target has been set for 95% of all SSSIs in England to be in Favourable Condition or Recovering Condition by 2010[4]. However, the most recent assessment of SSSI conditions in England by English Nature indicates that only 56.5% of assessed units (about three-quarters) are in Favourable or Recovering Condition (English Nature 2002). Furthermore, the condition of upland habitats within upland SSSIs are particularly poor. As indicated in Table 1.2, only about 30% of bogs, upland calcareous grasslands and upland heath were found to be in Favourable or Recovering Condition in 2002. The condition of upland acid grasslands was not much better.

[4] The target refers to the area of SSSIs

Table 1.2 The condition of upland BAP Broad Habitat types subject to burning in SSSIs in England in March 2001 (English Nature 2002)

BAP broad habitat	**% in Favourable or Recovering Condition as at 31 March 2002**
Bogs	32
Upland acid grassland	36
Upland calcareous grassland	28
Upland heathland	29

Note. The assessment is based on area of SSSI and excludes 3,892 units (18%), which had not been assessed by 31 March 2002.

English Nature reported that agricultural policy with respect to livestock was the most frequent factor constraining the attainment of Favourable Condition on SSSIs (affecting some 100,000 ha of assessed SSSI units). However, moorland burning was also considered to be a significant constraining factor affecting over 40,000 ha of assessed SSSIs.

2. Burning in the uplands

2.1 Fire as a natural component of upland ecosystems

Fire is a natural process, which most often occurs as result of lightning strike and affects many ecosystems (Wright and Bailey 1982, Rowe 1983, Moreno and Oechel 1994, Whelan 1995, Bond and van Wilgen 1996, DeBano *et al* 1998). In fact in many parts of the world it is, though perhaps less so recently, one of the principal processes that maintains open plagioclimax habitats, such as Mediterranean maquis (Trabaud 1994), lowland Atlantic heathlands (Webb 1986), savannah grasslands (Daubenmire 1968) and south African fynbos (Cowling 1992, van Wilgen *et al* 1992).

Despite this, as noted by MacDonald (1999), it is sometimes assumed that fire is a relatively recent impact on the British uplands (primarily being developed for game management from the late 19th century) and that natural fire is so unusual as to be almost irrelevant. However, although definitive information is lacking, there is certainly sufficient evidence to indicate that both natural and deliberately set fires have frequently occurred in many areas during the post- glacial period (ie Holocene). The extent to which natural fires have been a significant part of upland ecosystems in Britain has probably varied over time in relation to climatic conditions, the habitats present and the influence of man. Natural fires in the uplands may now be rare in Britain, but they are likely to have played a more important role in upland habitats in the past.

Post-glacial upland habitats in Britain initially consisted of rocky and tundra like habitats with scattered patches of *Salix* and dwarf *Betula*. But with increasing temperatures and rainfall, mires and moorland developed and woodlands spread up to a tree-line limit, typically between 400 – 900 m (Birks 1988). Thus, Birks suggests that at the time of the maximum extent of woodland cover in Britain, some 8,000 – 7,000 BP, there were extensive areas above the tree-line in North Wales, the Lake District and Scotland. But in contrast the Pennines, North York Moors and Bodmin Moor were probably mostly tree covered and much of Dartmoor was similarly forested. These upland woodlands were typically dominated by *Betula* and *Corylus avellena*, but with *Pinus*, *Quercus*, and *Ulmus* occurring locally in suitable conditions (Rackham 1976, 1986, Birks 1988).

There is some debate over the extent to which upland moorland and bog habitats developed naturally in Britain or were created by, or at least sped up by, later human activities. The balance of evidence, however, suggests that there was at least some natural moorland in high rainfall areas as well as at high altitudes and in the far north (Rackham 1986). Bog formation would have occurred naturally as a result of leaching under the cool moist climate of the uplands of Britain. This leaching would have led to the creation of base-poor soils, accumulation of acidic humus (mor humus) and the formation of humus-iron pans, which in turn would have inhibited soil drainage and aeration resulting in water logging and bog formation. Some blanket mires developed as a result of such natural soil processes as early as 9,000 BP (Birks 1975), whilst later climate change is thought to have initiated further blanket formation (Pennington 1970).

It is unlikely that natural fires occurred regularly in the mire habitats as wet ground burns rarely and may be considered as a fire refugium (Segerström *et al* 1996). But in contrast, natural fires are likely to have been an important process in upland forests, especially where there were conifers, as these trees burn readily. Evidence of the possible frequency and influence of fires in such former forests comes from studies of existing boreal forests in mainland Europe, where fire is an important part of their natural dynamics (Angelstam 1996). In these forests fires open up the canopy for generation, release nutrients which encourages plant growth and sometimes play an essential role in the germination of seeds. This helps to maintain a rotation between pioneer and climax stands in any one place, as well as the fertility of sites and the biodiversity of forests. In large-scale studies along river valleys in Swedish boreal forests, fire intervals were found to vary between 40-160 years, depending on vegetation type, topography and soil conditions (Kohh 1975, Zackrisson 1977a, b). The relationship between soil type and soil moisture has a particularly important affect on the flammability of vegetation and the probability of fire (Schimmel 1993). Under natural conditions lightning ignited fires in boreal forests occur approximately every 50 years on dry sandy and gravel sites, once every 100-150 years on moist sites and only once per 200 or more years on the wettest sites with the richest soils. Fire frequencies also vary with topography and aspect. Convex slopes burn more frequently than concave slopes, and south facing slopes burn more than north-facing slopes. Finally fire frequencies decline with increasing humidity (Zackrisson 1977a, Granström 1993).

Natural fire frequencies in boreal forests in Britain in the early Holocene may have been more frequent than those currently seen in Scandinavia as the climate was probably warmer and drier at that time (Briffa and Atkinson 1997), which would have increased the flammability of the vegetation (see Section 3.1).

There is also evidence for the past existence of frequent fires in Britain from the presence of charcoal in cores taken from lake sediments and peat bogs. For example, data from Scotland show that fires have occurred throughout the 11,000 year post-glacial period and that at many sites fires have been more frequent during prehistoric periods than during recent centuries (MacDonald 1999). Most paleoecology studies that include charcoal analysis do not have a sufficiently fine temporal resolution to determine probable fire return periods. However, a study of changes in the range limits of *Pinus sylvestris* some four-thousand years ago in Sutherland, used a high resolution stratigraphic study of a peat core and identification of fire scars as part of a dendrochronological analysis to estimate fire frequencies (Gear 1989, Gear and Huntley 1991). According to the charcoal stratigraphy study the average fire frequency was every 32.5 years, whilst the fire scar evidence suggested that fire intervals varied between 2 and 25 years. Thus, these results indicate that at that time fire return frequencies were not very different from those of much upland Scottish moorland today.

The uplands of Britain have now been dramatically changed by man, such that most of the upland woods no longer remain. This process of woodland destruction is likely to have started in Mesolithic times (c. 10,000 – 4,500 BC). For example there is evidence from blanket bog cores from Dartmoor of a continuous presence of charcoal between 7,700 – 6,300 BC, paralleled by a gradual reduction in arboreal pollen and the expansion of peat forming plants, heathers and grasses (Goodfellow 1998). This suggests that early settlers started to use fire to clear woodlands, probably initially as part of a strategy for attracting and hunting game, which led to an expansion of dwarf-shrub dominated moorland and grasslands at the expense of woodland cover. Such activities probably led to a suppression of closed tree cover above 350 m on Dartmoor, South Wales, the North York Moors, the Southern Pennines and possibly the Northern Pennines (Jacobi *et al* 1976, Birks 1988).

However, the main periods of deforestation in England occurred in the Neolithic period (4,500-2,000 BC) with a first pre-Roman Iron Age phase affecting most of England (except the Lake District) between 100 – 600 BC and then a second post-Roman phase between 1,400-1,700 BC affecting the Lake District (Jacobi *et al* 1976).

This replacement of woodland with heathland is likely to have increased the frequency and influence of natural fires as heathland vegetation is highly inflammable and will burn whenever the weather is dry (Rackham 1986, Webb 1986). Increased leaching of soils is likely to have taken place following the loss of tree cover and ground vegetation, which may have accelerated bog formation, particularly in the more southerly upland areas such as Exmoor, Dartmoor and Wales (Moore *et al* 1984).

2.2 The use of fire for vegetation management

2.2.1 Burning for livestock

By the end of the Iron Age (750 BC – AD 40 in England) all the large upland moors existed in Britain, although they were not as extensive as now and they would have been fringed by farmland and remaining natural woodlands (Rackham 1986). Since then English uplands have been primarily used for livestock grazing and burning is likely to have been used as a traditional means of maintaining the open vegetation and improving grazing conditions for sheep, cattle and ponies. Burning of upland moorlands for livestock has two principal objectives. Firstly, the removal of built up grass litter and encouragement of new nutritious growth (usually *Molinia*). The removal of the litter increases accessibility of the new growth to livestock, and the burning process releases nutrients that further stimulates growth. Where bare surfaces are created, then opportunities also arise for germination and temporary establishment of palatable grasses such as *Agrostis* spp. and *Festuca* spp (Currall 1981). Such burning management typically happens frequently, but burning can also be very effective in rejuvenating rough grasslands when ungrazed or undergrazed, and where, as a result, there is a large accumulation of dead material (Duffey *et al* 1974).

Secondly, it also helps to maintain the productivity of *Calluna*, which is the dominant plant in many upland habitats and (being evergreen) is an important source of winter forage for livestock.

In typical heathland conditions *Calluna* exhibits the following sequence of phases (Watt 1947, 1955):

- pioneer - the colonising phase, in which young plants may be associated with a variety of other species;
- building - the phase of maximum growth and productivity, with merging individuals and formation of a dense, even age canopy (the most competitive and exclusive phase);
- mature - the phase in which the canopy begins to become uneven and signs of gaps appear; and
- degenerate - the phase in which plants begin to die back from their centre, opening up obvious gaps.

The timing of the process varies considerably as it depends on growth rates which in turn vary according to environmental conditions. But typically the pioneer phase may last until the plants are 2 – 6 years of age, the building phase up to 10 – 15 years of age, and the mature phase up to 20 – 25 years of age, after which the plants gradually die back in the degenerative phase (Backshall *et al* 2001).

A key aim of burning is, therefore, to maximise the productivity of *Calluna* by preventing it from entering later phases of its growing cycle, when plants also become woody and unpalatable to browsing livestock. If carefully managed, moderate grazing of building-phase heather has the effect of keeping it in productive condition over an extended period, but in time plants tend to pass into the later phases (Gimingham 1995). This can be prevented by burning which rejuvenates the stand by vegetative or seedling regeneration. However, it should be noted that *Calluna* will not die-out without burning, and in many situations *Calluna* growth phases may be prolonged or even perpetuated. For example, layering may occur (especially in blanket bogs and wet heaths) through the development of adventitious roots as *Calluna* stems are covered by *Sphagnum* or humus (MacDonald *et al* 1995, MacDonald 1996).

Another aim of burning is to increase the accessibility of forage because much of the vegetation in tall stands of *Calluna* is inaccessible to, as they tend to avoid entering dense vegetation where their movement and view is restricted.

In practice, the overall aim of burning management for sheep is to maintain most heather stands below 20 cm height, but with some limited taller areas to provide accessible food during periods of snow (Backshall *et al* 2001).

In general moor burning was probably casual and sporadic until about 1800 (Rackham 1986). But in some areas regular burning dates back further. For example, on Dartmoor, livestock rearing has occurred extensively for the last 500 years and throughout this time burning is likely to have been a traditional management practice (Goodfellow 1998). This burning (or swaling as it is still known in the south-west) was primarily carried out on the commons and newtakes (moorland enclosed from the common a few hundred years ago) to achieve a flush of summer growth. It may also have been used to clear scrubby vegetation after periods of low grazing or management neglect.

Swaling has traditionally been carried out over large areas by simple means, with fires often being left to burn themselves out. Harvey and St. Leger (1953, cited in Ward *et al* 1972) considered that swaling was haphazard, there being a tendency to set fires to “a piece that will run”, thus resulting in fires that covered large areas. This was confirmed by Ward *et al* (1972) who examined swaling practices on Dartmoor in the late 1960’s by studying aerial photographs supplemented with sample field visits. They found that burning was commonplace on the areas of blanket bog (much of which was grassland dominated), *Calluna-Molinia* moorland and dry heath. The occurrence of past fires could be reliably detected for up to three years and they were able to map and measure the size of 108 fires that occurred between 1966-69. Analysis of their data (Tables 1-3) reveals that in each year the average fire size was 57.7 ha (1966/67), 37.3 ha (1968) and 29.0 ha (1969). Furthermore, 29%, 22% and 19% of fires were over 50 ha in extent respectively. Another examination of 107 fires known to have occurred on Dartmoor between 1965 and 1972, revealed that about one third were over 60 acres (Ward 1972).

The combined area of the blanket bog and heath vegetation, that is subject to regular burning on Dartmoor was estimated by Ward *et al* to be approximately 24,184 ha. The average total area burnt in each year was 1,471 ha, which suggests that burning return frequencies would be approximately once every 16 years. But this would depend on accurate recording of previous fire locations and careful planning and control of burns, which does not take place under typical swaling management. Thus Ward et al found from an examination of *Calluna* remains that actual fire return frequencies typically varied between 7-12 years (Table 2.1).

Table 2.1 The age of *Calluna* sample stands at the time of burning (between 1966-69) on Dartmoor (based on data from Ward *et al* 1972)

***Calluna* age class (years) at time of burning**	**Percentage in class (from 36 fires)**
<5 years old	0
5	5.6
6	0
7	16.7*
8	5.6
9	16.7*
10	8.3
11	19.4
12	13.9
13	5.6
14	5.6
15	2.8
>15	0

Note. * Some stands also had some older isolated clumps.

Thus swaling practices contrast with the normally smaller scale and usually more controlled methods of heather burning for grouse moor management (see below), of which there has never a history on Dartmoor.

Similar practices are also evident elsewhere, where burning is primarily carried out to improving grazing conditions for livestock. This is particularly the case on the uplands in the west of Britain, as the climate is generally too wet to sustain successful grouse moors. Large scale and frequent agricultural burns are typical of the south-western uplands of Exmoor and Bodmin Moor, parts of the Lake District and are well documented over much western Scotland (McVean and Lockie 1969, Currall 1981).

2.2.2 Burning on grouse moors

The regular use of fire to manage the heathlands of northern England and Scotland dates from about 1800 as a result of an increase in sheep grazing and the development of grouse shooting(Rackham 1986). As for livestock management, such burning is carried out to provide a supply of young, succulent grass and heather shoots, which are more palatable and nutritious for sheep and grouse (Moss *et al* 1972, Savoury 1978, 1986). But, unlike agricultural burns, traditional grouse moor management also aims to increase *Calluna* dominance and create a patchwork of *Calluna* stands of different heights and ages (Miller 1980, Hudson and Newborn 1995). In practice, the aim is to create mixed stands of mainly short pioneer and building phase heather (10-20 cm tall) for feeding, and some taller mature heather (20-30 cm) for nesting, shelter from predators and for feeding on during periods of snow.

Burning practices on grouse moors over the last 90 years have been considerably influenced by the findings of an inquiry into grouse disease and grouse moor management which analysed management practices on many of the most productive moors of the time (Lovat 1911). The inquiry's report recommended that the area to be burnt each year should depend on the time taken for the heather to recover and grow back to its desired height. As this varies according to soil and weather conditions it would typically result in a desired burning rotation (ie the time between fires) of 8 – 25 years. Thus the intention would be to burn on average 1/8 of the moor each year in the former case, or 1/25 in the latter case where heather regrowth is slow. The report also advocated the burning of heather moorland in very narrow strips (0.05 – 0.1 ha) to produce a complex mosaic of heather patches of variable age. However, it was recognised that even on the longer rotations it would be impossible to burn all the areas required in such small patches. Furthermore, the report also stressed that a desire to burn small patches should not take precedence over burning sufficient areas to maintain the required burning rotation. Lovat (cited in Phillips 1991a) therefore proposed the following elaborate management system to ensure that the necessary area is burnt whilst maintaining the patch system as far as possible:

1. Burn old heather in strips 50 yards wide [46 m], and let the strip run as far as the fire will take it.
2. Burn average 1.5 ft [0.46 m] heather in strips and patches of 0.25 – 0.5 acres [0.1 – 0.2 ha].
3. Burn patches and strips on the steep faces of the wintering grounds in small blocks of not more than 0.25 – 0.10 acres each [0.1 – 0.04 ha].
4. Burn the burnsides, knolls and nesting grounds of grouse in even smaller plots.
5. Burn the wet flow ground [bogs] in big patches of one to ten acres [0.4 – 4.0 ha].
6. Burn the high ground with northern exposure in large three-acre blocks [1.2 ha].
7. Burn good broad strips round each of the boundaries.
8. Treat specifically those portions of the moor which have a tendency to revert to grass.

Published information on actual sizes of burnt patches and fire rotations that have been achieved on grouse moors is scarce, but MacDonald (1999) provides an overview which indicates that the frequency of burning and the size of burnt patches have varied considerably over the last two centuries. Evidence from the inquiry on grouse disease (Lovat 1911) indicated that from 1800 to 1850 heather moorland was burnt in wide tracts by shepherds with about a tenth of the ground being burnt per year on average. But from 1850 to 1873 keepers became responsible for burning on many moors, and the extent and frequency of fires declined considerably, perhaps by a factor of ten. Burning subsequently increased, possibly to an average return period of about 30-40 years in the last quarter of the 19th century. Grouse shooting then declined in the 20th century, particularly after the second world war, which resulted in another decline in moorland burning.

Studies in Scotland suggest that over the last 50 years managed grouse moors have not been subject to as frequent burning as often assumed, with average burning rotations often being of 30-40 years or longer (MacDonald 2000). In a study of post-1945 aerial photographs of the Grampians and Borders region it was estimated that an average of 1-2% of heather had been consistently burnt per year, compared to the traditional ideal for grouse or sheep management of 6-10% (Hester and Sydes 1992).

Similarly, a study of post-1945 fires on Tulach Hill in Perthshire found that an average of less than 2% of the moor was burnt per year (Rhodes 1996, Stevenson *et al* 1996). But the number and size of fires varied over the period, from a small number of wide fires between 1944-50 to an above average number (10 per year) of very small thin fires from 1977-80, to a very small number of small fires between 1986-88. At other times, fire numbers were relatively constant and involved a mixture of sizes. These variations in fire size, did not appear to reflect changes in land use (sheep having been grazed throughout the study period), but rather the availability of labour and weather conditions during the burning seasons.

But it is important to note that consideration of the average area burnt each year can be misleading, as there may be substantial heterogeneity in fire history amongst different heather stands. For example, the Tulach Hill study found that many stands had not been burnt at all in the last 50 years, whilst a small percentage had been burnt up to six times (Table 2.2). This variation is likely to be fairly typical of large areas of moorland managed for grouse, or grouse and sheep (Rhodes 1996).

Table 2.2 Fire recurrence intervals for Tulach Hill heather stands over the period 1950-1988 (Stevenson *et al* 1996)

Number of times burnt post-1950	Percentage of land area
0	56
1	29
2	12
3	2
4	0.5
5	0.03
6	0.001

No comparable studies of actual sizes of areas burnt or burning rotations achieved appear to have been published for English grouse moors. However, it can probably be assumed that burning management has generally been more intense as game keeper densities have traditionally been higher on English estates and the extra manpower is likely to have enabled more burning to take place. In Scotland the optimal density of keepers was traditionally considered to be approximately one per 10-km^2 if shooting was let; but actual numbers were lower, varying from 0.6-0.8 10-km^{-2} until the end of the 1930's and then declining to about 0.5 10-km^{-2} since the Second World War (Hudson 1992). In contrast keeper numbers in England remained fairly constant at about 1.2 10-km^{-2} (although they were as high as 1.4 10-km-2 from 1920 to about 1940).

There appears to be little quantitative information in the published literature on the extent to which burning practices differed between upland habitat types in England, beyond the broad differences between burning for grouse management on heather moors (primarily in the north and east) and agricultural management on moorlands (mainly in the south-west and west). It is therefore not possible to quantify the proportions of specific habitat types that have been burnt nor their frequency of burning.

2.2.3 Current burning regulations and practices

Regulations

The burning of moorland and grasslands is regulated in England and Wales by the Heather and Grass etc (Burning) Regulations 1986 (SI 1986 No 428) as amended by the Heather and Grass etc (Burning) Regulations 1987 (SI 1987 No 1208). Any person who contravenes any provisions of these Regulations commits an offence under Section 20 (2) of the Hill Farming Act 1946, as amended by Section 72 (2) of the Wildlife and Countryside Act 1981. Other offences may also result from contravention of the Highways Act (1980), Environmental Protection Act (1990), the Clean Air Act (1993) and the Health and Safety at Work Etc Act 1974.

Key points that need to be considered when burning are summarised in (MAFF 1994) and Boxes 2.1 and 2.2.

As indicated in Box 2.2, there are special measures to control certain operations, on SSSIs under the Wildlife and Countryside Act, and these have been strengthened under the CROW Act. In England owners and occupiers of the land who wish to carry out any Potentially Damaging Operations (which are listed for each SSSI) and which would normally include burning, must give written notice to English Nature of the proposed operation. Written agreement to carry out the operation must be received unless four months has elapsed since written notice was given.

Box 2.1 Key advice on good burning practice for heather, grass, bracken, gorse and *Vaccinium*, from pages 2 & 3 of the *Heather and Grass Burning Code* (MAFF 1994).

BY LAW

Burning is allowed only between:

- 1 November – 31 March in lowlands;
- 1 October – 15 April in uplands[5]; and
- at other times under a licence that can be obtained only in very specific circumstances[6].

You must give not less than 1 day nor more than 7 days written notice of intent to burn to:

- neighbours; and
- owners and occupiers of the land.

You must not start burning between sunset and sunrise.

You must ensure that sufficient people and equipment are on hand to control the burn.

You must follow special arrangements and plan well in advance if burning on a SSSI.

You must not cause a nuisance through the creation of smoke.

You must not create dark smoke.

You must not start a fire which is likely to injure, interrupt or endanger road users.

REMEMBER

Plan

- Draw up a programme of essential burning on a sound rotational basis.
- Complete all burning outside the licensing and bird nesting periods.
- Scale individual burn sizes to manpower availability, safety requirements and weather conditions (check forecast).

Firebreaks

- Choose natural boundaries for the burn wherever possible, but if otherwise create firebreaks first as soon as possible in the season.

Control

- Have sufficient manpower and equipment.
- Appoint someone to be in charge.

Landscape and wildlife

- Take care of wildlife habitats.
- Take care of the landscape – especially woodland and hedgerows.

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[5] Severely Disadvantaged Less Favoured Areas

[6] Licence applications must be made to the local office of DEFRA

Neighbours

- Keep them informed and take account of their property interests.

Public safety

- Do not create hazards to road users and the public.
- Do not burn when the direction of the wind is likely to cause hazard or annoyance from the smoke or smuts.

The Law

- Know and comply with national requirements and restrictions.
- Know and comply with local bylaws.
- You can be fined heavily for breaches of the law.

Box 2.2 Advice relating to sites demanding special care aand attention because of their wildlife interest, from pages 6-8 of the *Heather and Grass Burning Code* (MAFF 1994).

SITES DEMANDING SPECIAL CARE AND ATTENTION

Do not burn

- ***Peat bog and wet moor (flow ground).*** There is little agricultural value to be gained from burning.
- ***Steep slopes*** where there is a risk of erosion.
- ***Scree slopes.***
- ***High altitude sites*** above the natural limit of forest (about 600 m or 2000 ft where growth is slow and the vegetation cover often patchy).
- ***Coastal heaths***
- ***Large areas of rank old heather.*** These burn too fiercely with poor regeneration. Mechanical cutting is usually better but if you must burn do so when the ground litter is still damp.
- ***Lowland heaths.*** These are particularly rich in flora and fauna and some species are rare (eg sand lizards and birds such as the Dartford warbler). The vegetation provides cover and food for animal communities (vertebrates and invertebrates) which undergo changes in phase with the vegetation. Usually there is little agricultural benefit from burning this type of vegetation and management by occasional use of machinery is sufficient to maintain any managed grazing already present.

Take great care when burning:

- ***Close to bracken.*** Heather should be cut for a distance of 5 m (6 yds) from the bracken edge. (A cutting implement should be used which will slice cleanly through the heather stems, rather than knocking them down). If this is not possible, burns should be at right angles to the bracken edge and should be narrow, no more than 30 m (35 yds) wide. Although slow, this avoids the risk of uncontrolled bracken spread.
- ***Near forests, woodland, scrub and hedges.*** Such areas are valuable in themselves and as cover for stock and wildlife. Consult the owner of the woodland concerned well in advance; they may wish to provide assistance in controlling the fire. Burning should take place only in ideal weather conditions and with full attention to safety factors such as firebreaks.
- ***Near or on Sites of Special Scientific Interest.*** These Sites contain important flora and fauna (or are of geological interest). Burning operations are invariably listed as damaging operations under the Wildlife and Countryside Act, 1981. Consultations with the local office of English Nature or the Countryside Council for Wales is essential in these areas.

There is similar regulation of muirburning in Scotland, where it is primarily governed by the Hill Farming Act 1946. These regulations are summarised in *The Muirburn Code* (Anonymous 2001b) and useful background information on fire behaviour and its effects is provided in an accompanying booklet *Prescribed burning on moorland. Supplement to the Muirburn Code: a guide to best practice* (Anonymous 2001c). Both are obtainable from the Scottish Executive web page at http://www.scotland.gov.uk/library3/environment/mbcd-00.asp

Burning in Scotland is permitted between the 1st October and 15th April inclusive below 450 m (1500 ft) altitude. This may be extended to 30th April on the authority of the proprietor or of the Scottish Executive Environment and Rural Affairs Department (SEERAD). In Upland regions, ie above 450m, burning may take place from the 1st October to the 30th April, which may be extended to the 15th May as above. Generally SEERAD does not encourage the extension of burning periods, but the proprietor does not need the permission of SEERAD for extension periods. Unlike in England and Wales, there are no provisions for further extensions before or after these dates, by either the proprietor or the Scottish Executive.

Although the legal burning season appears quite long (ie over 6 months in upland areas), in practice the time over which burning can be carried out is usually relatively short, primarily because weather conditions are often unsuitable (Currall 1981, Hobbs and Gimingham 1987). Vegetation may be too wet to ignite or covered with snow for much of the season in many upland areas. In dry windy conditions, even in winter, vegetation can dry out quickly but wind speeds must not be too great to risk losing control of the fire. Another limitation is that burning must also be carried out during daylight, and days are relatively short during the burning season. Thus the correct combination of conditions may only occur a few days in each season (Miller 1980). In practice, burning therefore seldom takes place in winter and is usually restricted to a short period in spring and a few days in autumn. Though when conditions are suitable, such as in spring 2003, burning may be carried out over many weeks.

There appears to be no published information on the extent to which burning in English uplands takes place within the prescribed periods. DEFRA do not compile records of reported incidents of illegal burning outside prescribed period because it is difficult to reliably identify the cause and intention of most fires, some possibly being accidental or arson (DEFRA pers comm 2003). However, specially licensed burning outside the normal legal period does appear to be uncommon. According to DEFRA, typically five to seven license applications for burning outside the normal legal period were received and granted annually in England between 1993 and 1998. In 1999, the latest year with compiled data, 9 applications were received and approved.

Recent burning management practices

Prescribed burning remains a common practice in much of upland Britain, primarily for agricultural and/or grouse management purposes as described above.

Such burning, when carried out carefully and in appropriate habitats, may also provide secondary biodiversity conservation benefits (whether intended or otherwise) by preventing succession and enhancing structural diversity. In limited specific circumstances, controlled burns may also be used for fire control purposes, eg by creating firebreaks or limiting the height and quantity of woody vegetation, thereby reducing the potential intensity of any fires that arise.

There appears to be few statistics available on the current extent of burning of different habitat types in England. However, it can be inferred from the uses to which upland habitats in England are put, and general literature, that burning in the uplands is principally carried out on dwarf-shrub heaths, blanket bog and rough grasslands, particularly south-west grasslands that are now dominated by *Molinia* (much of which is degraded blanket bog). Probably for practical reasons, burning appears to become less usual as habitats become higher and wetter, although some burning of the wetter heaths and bogs also certainly occurs. Upland woodland, calcareous grassland and enclosed improved grasslands are not usually burnt, except as a result of accidental fires or arson.

The frequency of burning and the amount of planning and care with which fire management systems are implemented varies amongst habitats and from place to place (Hobbs and Gimingham 1987). Prescribed and controlled management burning is mostly carried out on dry heathlands that continue to be managed as grouse moors. Within England, grouse moor management remains a major landuse over much of Northumberland, the Pennines, the Forest of Bowland and the North York Moors (Brown *et al* 2001).

Current thinking on grouse moor burning practices is that, on all but the most intensively managed moors, Lovat’s (1911) recommendations for fire sizes of 0.05 – 0.1 ha are much smaller than they need to be (Phillips 1991a). The Heather Trust now suggests that fires should ideally be about 30 m wide and as long as practical (Straker-Smith and Phillips 1994), which would give a fire area of 0.3 ha if 100m long. This is based on research by the Banchory Grouse Team that showed that territorial grouse rarely move much more than 15 m from good escape cover. On poor ground where lower grouse densities may be expected they suggest that there may be no serious disadvantages to grouse in having fires of up to 50 m width (ie 0.5 ha if 100 m long). Fires of 30 m width are also recommended by MAFF and the Scottish Executive (MAFF 1994, Anonymous 2001c).

Circumstances affecting the management of grouse moors have also changed considerably since the inquiry report *Grouse in health and time* was published (Lovat 1911). In particular grazing pressures have increased considerably as a result of higher sheep stocking rates, and in Scotland the presence of three to four times as many red deer (Phillips 1991a). This has been further exacerbated by a reduction in sheparding of hefted flocks. As a result large areas of moor need to be burnt each year (especially when burning programmes are being re-established) to ensure that sufficient young forage is provided to avoid concentrations of livestock and overgrazing of regenerating growth (Phillips 1991a, Anonymous 2001c).

Another important factor affecting current moorland burning practices is the reduced availability of inexpensive casual labour to assist with burning management. This may be significantly reducing the capacity for carrying out large numbers of small burns on many estates. However, this may be offset by technical advances and new equipment, such as the long-handled lightweight metal beater (and face mask), which has transformed an individual’s effectiveness in fire control (Phillips 1991a). Tractor mounted swipes may also now be used now to cut firebreaks and water jet sprayers give further means of control. Overall it is considered that a team burning in open heather swards under reasonably dry conditions with good beaters and no mechanical assistance will do well to achieve 4 ha per day if 30 m wide strips are burnt (Straker-Smith and Phillips 1994). But on ground where a swipe or water jet can be used, or where there are numerous natural fire breaks (eg wet ground) or recently burnt areas, then burning can be carried out at two or three times this rate.

But again information on what is actually being achieved in terms of burning areas and frequencies on grouse moors is scarce, particularly in England. The recommendations described above suggest that most burns are likely to at least aim for small patches of 0.3 – 0.5 ha. But in practice many fires end up being much larger and there are recorded cases of some that get out of control and cover 100’s of hectares particularly during periods of dry settled weather (English Nature pers comm). Actual fire return frequencies are also largely unknown on English grouse moors. Some evidence from the North York Moors suggests that there has been a recent decline in the practice of rotational burning in recent decades as sample heights of heather in 1982/83 indicated that 27% was over 40 cm high, and 82% of this was in the mature or degenerate phase (Leathard 2001). Although no data are provided on previous *Calluna* heights, Leathard concludes that the high proportion of mature and tall heather is "likely to be the result of a decline in the practice of rotational burning in the preceding decades". However, it is not known to what extent this is representative of other areas of heather moor in England. The National Park has subsequently encouraged burning through incentives and its Moor Management Programme and this appears to have increased burning. 1992 survey results indicated that heather height had been reduced as only 43% was more than 25 cm high and only 13% was over 40 cm (Leathard 2001).

As noted above, traditional burning practices in the south-west uplands differ from those in the northern uplands as, in the absence of grouse shoots, fires have been primarily set by graziers solely for agricultural purposes. Current burning practices appear to differ little from traditional swaling, at least on Dartmoor (Goodfellow 1998). On Dartmoor all types of moorland vegetation are burnt regularly, including *Calluna*, *Ulex* spp. *Molinia* and *Pteridium* stands. *Molinia* and *Pteridium* are burnt about every three years, whilst *Ulex* and *Calluna* are burnt about every seven years (Goodfellow 1998). In the newtakes, the tenants are controlled to some extent by their landlords (usually the Duchy of Cornwall) and by management agreements with the Dartmoor National Park Authority. Burning in these areas is normally carried out carefully and in rotation for conservation and agricultural aims. However on the commons (which encompasses the SSSI and candidate SAC land) the situation is more varied, with some large areas being burnt repeatedly and with little control (Smallshire *et al* 1997). For some time there have been indications that burning has been increasing and too frequent, possibly causing a decline in *Calluna* cover and the loss of heath to bracken (Ward 1972, Ward *et al* 1972, Wolton *et al* 1994).

Large fires still appear to be the norm on Dartmoor, as it is reported by RDS (2002) that fires range from 0.5 hectares through to many hundreds, or even in some instances thousands of hectares where fires have got out of control. There have, however, been recent initiatives to encourage the burning of smaller fires.

As noted above, the legal requirement is that burns must be carried out in the uplands between 1 October and 15 April. The Dartmoor National Park recommends no burning after 31 March and few burns are carried out after this (RDS 2002). Due to the requirement for suitable weather, burns are mainly carried out during the last few days of March, and occasionally in February and October.

There has been pressure in recent years to reduce the burn period on Dartmoor to between 1 October and 15 March to offer greater protection to early breeding birds. However, agricultural interest would prefer a longer period than presently recommended.

On Exmoor, heather is burnt periodically and like Dartmoor often over large areas, but a study in 1984 found that fires are properly planned and controlled (Miller *et al* 1984). In contrast, at least at the time, all the grass moor (which was mostly under one estate) was burnt each spring. This was to remove the dead grass and encourage fresh growth for the lambing season in April and May.

2.2.4 Cutting as an alternative to burning

Cutting heather has been considered as an alternative to burning as it has various potential advantages. Firstly, it might be expected to produce better results than burning, since a greater number of sprouting centres are retained (Mohamed & Gimingham 1970). It is also much less constrained by weather conditions. It may be appropriate for the creation of fire breaks and where burning is not possible (eg because of the proximity of forests or houses) or where there is insufficient labour to control fires. It may also be preferable where burning may damage habitats or species of high conservation value (Rowell 1990).

According to the supplement to the *Muirburn Code* (Anonymous 2001c), the most practical technique for heather cutting is to use a chain swipe mounted on a four-wheel drive 80 or 100 HP tractor, which can be fitted with double wheels for softer ground. This produces finely mashed up cuttings which appear to decay quite rapidly, especially in wet conditions where an alternative to burning is most needed. It is particularly valuable where there is much *Molinia* among the heather. On ground suitable for the use of a double-chop forage harvester, the cuttings can be blown out side the cut patch. Cuttings can also be baled and in some cases sold for mulch (eg to restore burnt or eroded areas) and filtration uses.

However, studies of young *Calluna* stands in the North York Moors National Park found that in practice there was little difference in the extent of regeneration between cutting and burning regimes (North York Moors National Park 1991), and burning seemed to be more successful for older stands, probably because regeneration was more reliant upon seedling establishment. In dry conditions the cuttings may not decay quickly and may form a thick mat which suppresses regeneration.

There are also practical limitations that restrict its use as a management technique. In particular cutting is not feasible on stony, boggy or steep ground, which therefore precludes its use on much of the uplands. Cutting machinery should also not be used where there are archaeological remains as these may be damaged.

Cutting is also a more expensive method of rejuvenating *Calluna* in terms of time and equipment than burning (North York Moors National Park 1991), although this may be partly compensated for by using or selling the cuttings. The capital costs are high and operational costs per unit area, using a contractor, is likely to be about 20% more costly than burning, with cut firebreaks and foam traces, which in turn is about 70% more costly than traditional controlled burning (Anonymous 2001c). However, if labour and suitable machinery are already available then cutting or swiping can be done for about half the cost of traditional controlled burning.

2.3 Accidental fires and arson

Wildfires are uncontrolled fires, which commonly occur in much of the British countryside, and particularly within the National Parks. These may occur as a result of prescribed management fires getting out of control (eg when fires are poorly planned, set in windy conditions or with insufficient firebreaks or labour to restrain them). But many damaging wildfires arise from accidents associated with recreational uses (Anderson 1997). They typically start close to roads, paths and other access areas, often originating from discarded cigarettes or sparks from barbeques and open fires. Most happen in the summer, when weather conditions are dry and warm, because this both increases the fire risk and the numbers of visitors present (Anderson 1986).

Arson also commonly occurs for various reasons and has led to some major wildfires over large areas, eg on Trendlebere Down SSSI on Dartmoor.

As described above, this report focuses on examination of the impacts of prescribed burning on habitats and species. However, studies have shown that when accidental or deliberately set wildfires occur, such as in the North York Moors National Park in 1985 (Maltby 1980, Maltby *et al* 1990, Legg *et al* 1992), they can be particularly damaging to habitats and their associated fauna (eg nesting birds). In dry conditions they may be very intense, causing severe damage to soils and ignition of the underlying peat, which may then burn for weeks (see Section 4.1.1). Such peat fires may cause long-term and, in some cases irreparable damage. Studies of such wildfires and their impacts are therefore reported in this review where they provide information that is relevant to consideration of the impacts of large poorly controlled management burns. It is also necessary to consider the impacts of wildfires as managed fires have an important role in their prevention and mediation.

3. Fire characteristics

To understand the impacts of fire on habitats and species it is necessary to have at least a basic understanding of the physical properties of fires and the factors that influence them. Fire characteristics have, therefore, been the subject of considerable research, much of which has been carried out on upland heathlands, but additional relevant information also comes from some studies of lowland heaths and other habitats. An outline of the key factors affecting the character of fires is provided below, which draws on this scientific research and summaries provided in the supplement to the *Muirburn Code* (Anonymous 2001c) and Hobbs and Gimingham (1987). A simplified illustration of the interrelationships between the key factors determining the character of a fire is provided in Figure 3.1.

3.1 Flammability

A fire ignites when the temperature of a potential fuel is raised in the presence of oxygen above a critical temperature, which for most vegetation is between 325 – 480 ^{0}C (Pyne *et al* 1996). At this point chemical reactions in the fuel generate more heat than is absorbed and self-sustaining combustion begins. However, the ease of igniting the fuel is greatly influenced by a number of factors.

Of most importance is the moisture content of the potential fuel, because water requires a large amount of heat to raise its temperature. Dead plant material dries out most quickly and burns readily, and therefore is usually the fuel that ignites first and then sustains a fire. The moisture content of dead grass leaves (such as *Molinia*) can change from 90% to 20% in less than half an hour under drying conditions. In contrast, green foliage of deciduous plants often has a high moisture content (up to 300% or more of dry weight) and is consequently very difficult to ignite (Anonymous 2001a). But heather (as well as conifers and some other evergreens), often have lower moisture contents and contain volatile compounds that burn readily. Normally, there is considerable variation in moisture content within the vegetation and soil. For example, Hobbs and Gimingham (1983) found that in March and April vegetation moisture levels varied between 49 - 144% (% of dry weight) and soil moisture levels ranged between 45 - 305%. However, on dry, well-drained ground, moisture contents of the vegetation and litter may fall below 50% and loose plant litter supported among the shoots of the heather canopy may have a moisture content of only 25% (Anonymous 2001c).

The structure of the potential fuel also influences the speed of ignition and spread of a fire. The fineness of the material, the ease with which air can mix with it, and how uniformly it is distributed are the most important factors. Fine material less than 6 mm diameter (such as the leaves and twigs of young heather) dry more quickly and completely, ignite more easily, and burn more quickly, than large twigs or stems.

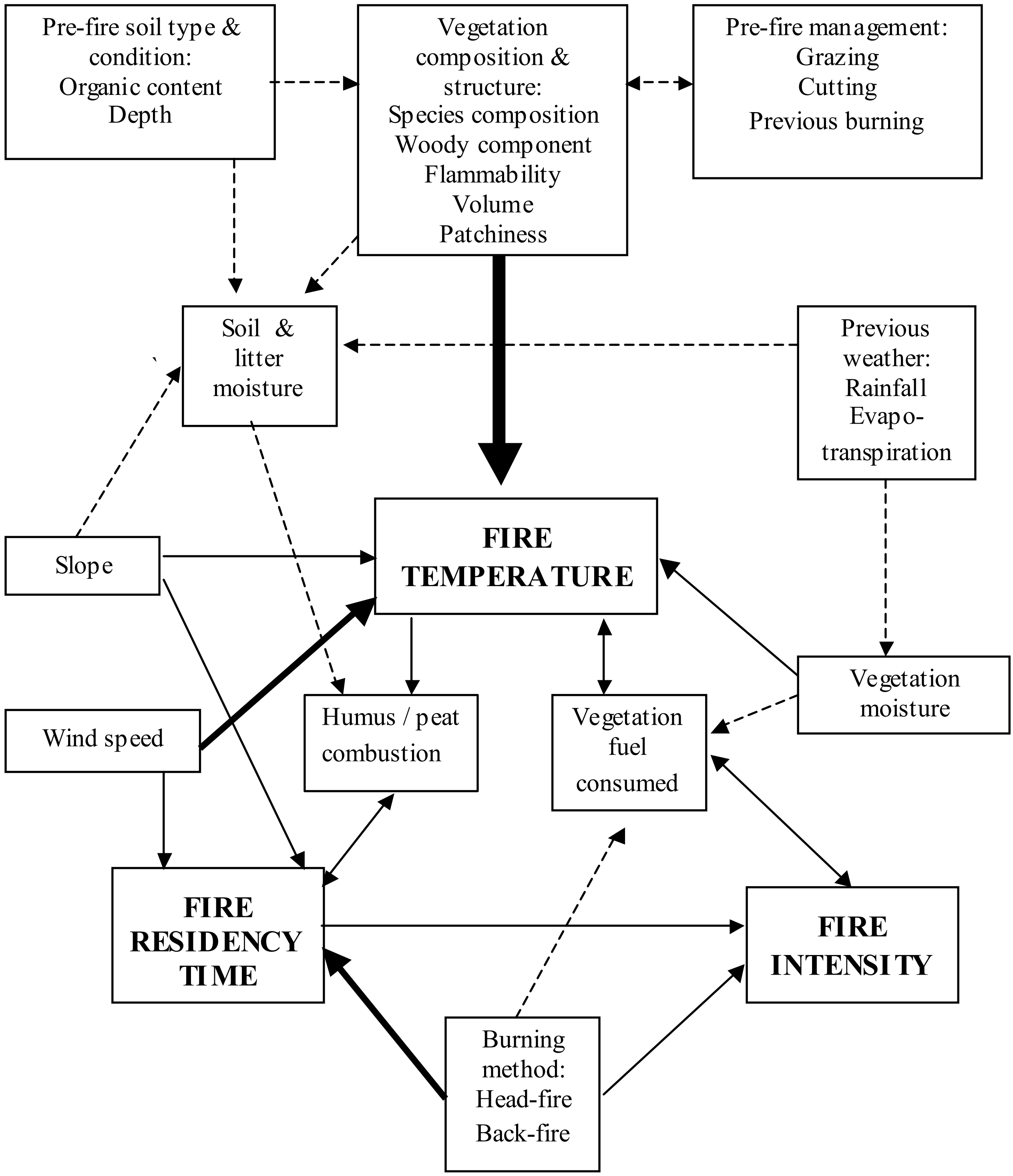

Figure 3.1 The key factors determining the character of a fire (in terms of temperature, intensity and residency time) and their interrelationships.

Line widths are indicative of the approximate importance of the factor. Dashed lines indicate indirect influences.

3.2 Fire temperatures and intensity

As described further below, the temperature and intensity of a fire is one of the key factors affecting its impacts. Fire intensity is most commonly defined as the rate of energy or heat release per unit time and per unit length of fire front, which actually equates to fireline intensity, first defined by Byram (1959). Fireline intensity is particularly important from a fire fighting point of view, but of greater ecological importance is likely to be the total energy released per unit area at the ground surface, which is hereafter referred to as ground surface intensity. Measurements do not normally appear to have been made of ground surface intensities in fire studies, but in most situations they are likely to be positively correlated with fireline intensities.

Fire temperatures and fireline intensities have been measured in heath fires in north-east Scotland using Thermocolor heat-sensitive paint (Whittaker 1961) and thermocouples (Kenworthy 1963, Kayll 1966, Hobbs and Gimingham 1984b), and in lowland heaths using thermistors (Webb 1986). These studies found that fire temperatures vary considerably, depending on how and where in the vegetation it is measured, and factors affecting the intensity of the fire. The study by Kayll (1966) suggested that temperatures may only reach 250 ^{0}C in the canopy, but this may be atypical as only 27% of the available fuel was consumed. Other studies recorded fire canopy temperatures of 220 – 840 ^{0}C (Whittaker 1961), 940 ^{0}C (Kenworthy 1963, cited in Webb 1986) and 340 – 790 ^{0}C (Hobbs and Gimingham 1984b).

Hamilton (2000) studied fire characteristics in blanket bogs in north-west Scotland and recorded mid-canopy temperatures of 31 – 886 ^{0}C for all plots and 103 – 886 ^{0}C for all plots containing *Calluna*. Although there are methodological differences between this and the earlier studies on drier heaths, the results indicate that temperatures can get as high as on dry heaths, but there appears to be a greater proportion of low temperatures in bog fires. Hamilton considers that the main reason for the high variability is the spatial patchiness of the fuel and to a lesser degree the extreme moisture gradients in the fuels. Currall (1981) measured temperatures attained in two experimental fires (using Thermocolor pyrometers) on wet heaths on Skye. One plot, with 20 cm high *Calluna* and wet ground conditions, produced a fire with a maximum temperature of 670 ^{0}C, with 50% of readings over 450 ^{0}C. But this did not produce a clean burn and some *Molinia* litter remained. The second plot, had *Calluna* of 30 cm high and moist ground conditions. This produced a cleaner burn with a maximum temperature of 800 ^{0}C, and 50% of readings over 500 ^{0}C. This burn therefore produced comparable temperatures to those observed on dry heaths.

Thus, variability is an important feature of all fires, but this appears to be particularly the case with fires on blanket bogs and wet heaths, which demonstrates that fires cannot be treated as a uniform event.

One study has been found that measured temperatures in grass fires (Lloyd 1968). In this study Lloyd used heat-sensitive paints to measures temperatures in experimental fires, in two years, in 2 x 2 m *Festuca-Helictotrichon* grass plots in the Derbyshire Dales. The measured temperatures varied considerably between tussocks and open soil areas. Tussock temperatures ranged mainly between 560-640 ^{0}C in one year and 715-805 ^{0}C in a second drier year. Soil surface temperatures were below 100 ^{0}C in open areas, but up to 340-440 ^{0}C close to tussocks. Despite some potential inaccuracies with the use of heat-sensitive paints and the small size of the fires (which reduces temperatures – see below), these results suggest that grass fires, at least in these habitats and conditions, have comparable temperatures to those in dwarf shrub heathlands.

In most fires very high temperatures only last for a short time, rising and declining rapidly as the fire front passes. The duration of ambient temperature rise was found to be only 30 – 60 seconds by Kenworthy (1963) and less than one minute, except perhaps in grass tussocks, by Lloyd (1968). Hobbs and Gimingham (1984b) found that ambient temperatures were raised for 5 minutes, but the time above 400 ^{0}C was typically 30 – 60 seconds, and about 30 seconds for temperatures above 600 ^{0}C.

There can be considerable differences between canopy and ground temperatures, and in some cases raised temperatures persist for longer at the ground surface, but they are generally similar (Hobbs and Gimingham 1984b). However, if much of the fire is carried in the crowns of tall vegetation then the ground surface temperature (and ground surface intensity) may be relatively low as heat transfer by radiation declines rapidly with distance (ie proportional to the square of the distance).

Even when ground temperatures are high, soil temperatures appear to be little affected by most fires as moss and the upper layers of litter provide extremely good insulation. Few measurements have been made of soil temperatures during fires, but Hobbs and Gimingham (1983) found that the highest temperature recorded at 1 cm depth in freely draining podzolic soils was 70 ^{0}C. Web (1986) observed similar soil temperatures in lowland heath fires and found no detectable increase at all at 4 cm (but does not provide details of the soil type). Lloyd (1968) measured only a 3 ^{0}C rise at 0.5 cm depth in a well drained rendzina soil as a result of a grassland fire.

The extent to which the soil is warmed depends very much on its moisture content. This is because temperatures in the soil cannot rise above 100 ^{0}C until all water has evaporated, and this is unlikely to occur unless high temperatures are maintained for a long period (Roberts 1965, Cromer and Vines 1966). No measurements of soil temperatures during fires appear to have been carried out on peat soils, but it might be predicted that under saturated conditions they would provide effective insulation given their high water content.

Most importantly, the impacts on soils depends on how much, smouldering combustion occurs on the ground. In moorland management fires, the fuel that is not completely burnt by the flames generally cools rapidly and smouldering is sustained for no more than a few minutes. However, the ground surface is less exposed to heat loss, and if peat is present and ignited then this may smoulder for days or even weeks (Johns 1998). If the peat ignites, temperatures of 600 ºC can be reached and temperatures greater than 300 ºC may persist for hours or even days where smouldering continues (Anonymous 2001c).

There are a number of factors that have been found to affect the temperatures of fires. In particular, the amount of heather present in moorland vegetation is often a principal determinant of the temperature and intensity of a fire, but the structure of the vegetation is also important. Kenworthy (1963), found that fire temperatures appeared to increase in a simple linear relationship with stand age. But the detailed studies by Hobbs and Gimingham (1984b) showed that temperatures and fire intensity increase with stand age up to the late mature phase of *Calluna* development (ie 20-25 years), after which the temperature and intensity of the fire declines, but variability increases. The duration of high temperatures also increases with stand age and only declines in the degenerative phase.

Overall Hobbs and Gimingham found that maximum canopy fire temperature could be relatively well modelled by a multiple regression of vegetation height, fire width and wind speed, where:

Max temperature = 11.10 vegetation height + 12.79 fire width – 3.61 wind speed

(R = 0.87, P <001; contributions to R^2: vegetation height = 0.59; fire width = 0.54; wind = 0.14).

Vegetation height was in fact mainly a function of vegetation age and in turn an integrating factor describing the structure of the stand and its fuel availability.

The fact that fire temperatures are positively correlated with fire width (and explains nearly as much variation as vegetation height) has important management implications. As a result, lighting patterns (eg as a series of spot fires or as a continuous line of fire) can considerably influence the behaviour of a fire and the distribution of fuel can produce similar effects. A fire in vegetation that has a patchy distribution of burnable material will have some of the characteristics of a series of small individual fires rather than a single, continuous fire front.

There is some debate over the effect of wind on fire behaviour and temperatures. It has been suggested that high wind speeds fan fires and thereby increase fire temperatures (Whittaker 1961, Daubenmire 1968). However, Hobbs and Gimingham (1984b) found that wind speed only accounted for a small amount of the variation in temperature. Furthermore, it had a negative effect, indicating that it may cool the fire or lead to less complete combustion as the fire spreads more quickly. They also report that similar effects were found in pine woodlands in Canada (Sparling and Smith 1966) and *Populus tremuloides* woodlands in Canada (Smith 1978).

Byram (1973) found a marked decline in heat yield with increasing vegetation moisture content. But moisture content was only found to be of secondary importance by Hobbs and Gimingham (1984b). The probable reason for this is that the management fires studied took place within a fairly limited range of moisture conditions. Wetter conditions would not enable ignition of the vegetation, while burning would have been avoided under drier conditions to avoid the possibility of lossing control of fires. Nevertheless, variations in the moisture content between different vegetation components may be important in causing variations in fire conditions from place to place, and especially in terms of affecting heat transfer to the soil.

Hobbs and Gimingham (1984b) also calculated fireline intensities (ie the energy released as heat per unit time and per unit length of fire front) for one site and found that these varied between 43 – 1,112 kW m^{-1}. Fire intensity varied according to stand age (which relates to the amount of available fuel), as with fire temperature, but also rates of spread. Fireline intensities in blanket bogs in northwest Scotland were found to be highly variable but of much lower intensities, ranging from 11 to 120 kW m^{-1} (Hamilton 2000), despite the fact that maximum temperatures reached were similar to those in other heathland fires.

3.3 The shape and speed of burns

The shape of burns is often dictated by physical barriers or boundaries and by slope, whilst the speed of a burn is mainly controlled by wind strength and slope, and to an extent by human influences. Timing of the burn will also have an influence, for example, if it is carried out after a period of damp or dry conditions.

Most moorland management fires are set to burn with the wind, and are termed 'head-fires' or 'heading fires' (Hobbs and Gimingham 1987, Anonymous 2001c). This allows the wind to bend the flames forward over the unburnt vegetation which pre-heats and dries it, and therby facilitates ignition and fire spread. Compared with fires burning against the wind ('back-fires' or 'backing fires') head-fires spread much more quickly, produce longer flames from a wider flaming zone, and have a greater intensity but consume the fuel less completely. Back-fires have the opposite characteristics and are little affected by wind speed. However, the most important feature of back-fires is that heat production is concentrated closer to the ground and as a result they consume more of the fuel and produce "cleaner" burns (Anonymous 2001c). Back-burning may be used for burning old *Calluna*, on wet ground, or for creating clean firebreaks but it is too slow, and potentially too damaging to the vegetation and soil, for very extensive use unless used very carefully. The properties of a flank-fire are intermediate between those of a head-fire and a back-fire.

Burning up slopes tends to produce rapid and intense fire because as it burns it pre-heats and dries the vegetation above it. As a fire moves onto a slope, it will tend to swing upslope and its rate of spread increases; doubling for every 10º increase in slope.

According to the supplement to the *Muirburn Code* (Anonymous 2001c) a well-controlled head-fire in relatively uniform 20 – 30 cm heather tall, in recommended weather conditions, will move forward at about 0.5 – 5 m per minute, being fastest with dry conditions and some wind. Fires may spread at up to 8 m per minute with head-fires in heavier, drier fuels or with relatively high wind speeds. Back-fires spread much more slowly, normally at between 0.3 – 1 m per minute. Smouldering fires in peat move at rates of only about 3 cm per hour.

3.4 The characteristics of wildfires

The discussion of fire characteristics has been largely restricted to conditions observed in prescribed management fires, primarily because this is the focus of this study, but also because of the practical difficulties of obtaining measurements from unpredicted fires. However it is worth noting that wildfires will be typically much more intense than those observed under managed conditions because they usually occur in particularly dry conditions (as the highest risk is in summer), often affect areas of tall woody vegetation and cover wide areas. Few measurements have been made of wildfires, but it is interesting to note that an intensity of 2,430 kW m^{-1} was calculated for a heath fire in Scotland (Kayll 1966), which is much greater than the range measured for managed fires by Hobbs and Gimingham (1984b), the maximum of which just exceeded 1,100 kW m^{-1}.

With dry substrates, such fires are much more likely to ignite humus or peat layers as well as the vegetation, which may then burn for considerable periods (Rowe 1983, Johns 1998). This may cause substantial damage which in turn may result in poor vegetation regeneration and excessive prolonged erosion (see Section 4.1.1 and 4.3).

Rates of fire spread can also be much greater in wildfires as these may take place when the weather and fuel conditions are outside normal recommended burning conditions. In such circumstances wildfires can move at speeds of 50 m per minute or more (Anonymous 2001c).

4. Physical and chemical impacts of burning

4.1 Soils

4.1.1 Physical impacts

Fire can have a number of physical effects on soils (which here includes mineral soils, peat and humus), including changes in micro-climate, structure, porosity and hydraulic conductivity. The most severe initial impacts of burning may be the ignition and consumption of peat and humus layers, which as noted in Section 3.2 can be the result of intense fires or slow smouldering fires in dry conditions. One such wildfire took place on 600 ha of Rosedale Moor on the central plateau of the North York Moors in the hot and dry summer of 1976, and the physical impacts of this have been studied in some detail (Maltby 1980, Maltby *et al* 1990). The effect of the fire was relatively slight on the areas of mineral soil, but the fire burnt deeply into some areas of blanket peat and the organic horizons of some soils. This resulted in substantial physical and chemical changes to the peat and soil materials. The shallowest peats (200-400 mm) which had dried out during the summer were entirely consumed and reduced to ash. Deeper peats, between 400-800 mm were mostly severely charred, and in many areas formed distinctive contracted columnar peat structures. Where the blanket peat was sufficiently deep and remained wet (mostly > 1 m deep and with a water content of over five times its dry weight) the peat was little affected by the fire, although the surface vegetation, litter and partially decayed humus were removed. Wet flushes suffered little direct effect from the fire. Importantly for the post-fire recovery of the site, charred, non-fibrous humified peat formed crusts which were easily detached (for example by needle ice) and then readily removed by water erosion. Bare peat also broke up into granules, creating a surface that was susceptible to wind and water erosion, and insufficiently stable for plant recolonisation.

Another example of a severe summer wildfire was on Howden Moor on the Pennines where an accidental fire in August 1959 burnt for 28 days and consumed about 1 m depth of peat (Tallis 1973). Even 10 years later up to 80% of the ground remained as patches of bare peat in some areas.

Soil loss can lead to profound effects on the ecological character of an area. In particular, where fires consume peat layers overlying mineral soils, then the peat land ecosystems will be essentially destroyed and will take many centuries to regenerate, if they ever do. One example of similar soil impacts comes from a fire on the plateau edge of Coombsdale, in the Derbyshire Dales. This destroyed the vegetation and humus of an acid, highly organic limestone ranker, reverting it to a rendzina and thereby radically altered the vegetation from acid grassland to limestone grassland (Grime 1963).

Severe fires may also change the physical properties of the surface layers of the remaining soil. On peats this may include the formation of a surface crust of tarry bitumen as a result of the ignition of peat waxes in intense fires (Clymo 1983). Although this may help to reduce erosion of the underlying intact peat it is likely to increase water run-off which will exacerbate the erosion of surface ash and therefore nutrient loss (see next section). It may also retard colonisation by *Calluna* and other desirable plants, thereby prolonging the period without vegetation cover. Water-repellent compounds may also be deposited within the soil by distillation during prolonged smouldering fires. These may create layers in the soil that interfere with water and root penetration, and may create structural weakness (Anonymous 2001c).

In less severe fires, ie well controlled management burns, the substrate is not usually ignited and the main immediate impact is the removal of some or all of above ground vegetation, leaving an area of blackened ash-covered exposed soil or humus. As a result of the loss of shading by vegetation, there are greater fluctuations in soil surface temperatures and moisture regimes, with increased temperatures and risks of desiccation in summer (Mallik 1982, Tallis and Yalden 1984, Salonen 1992). The higher daytime temperatures may stimulate vegetation growth but at the same time they create inhospitable conditions for seedlings which may die from drought. Temperature ranges in the upper soil are generally increased, on both diurnal and seasonal timescales, and the formation of needle ice is an important process leading to detachment of particles from bare peat and soil surfaces (Maltby *et al* 1990).

Much of the ash from a fire may be quickly eroded (see below), but at least some is washed into or otherwise incorporated into the soil, which leads to further physical and chemical changes (see Section 4.2 for discussion of effects on nutrients). Mallik *et al* (1984a) found that infiltration rates on freely drained brown podzolic soils declined by 74% on burnt heather plots compared to unburnt plots, probably as a result of ash particles clogging the soil pores in the upper surface layers. However, they note that other possible causes of the reduced infiltration may have been changes in water repellancy, soil bulk density, thermal conductivity, evapotranspiration rates, hysteresis and soil micro-organisms. Their results concur with similar studies in other countries, but it is noteworthy that a similar study of heathland burning in Scotland by Kinako (1975) found a 30% increase in infiltration. Mallik *et al* suggest that Kinako's method of measuring infiltration may have been unsatisfactory, as it was unsuccessful when tried in their study.

Mallik *et al* (1984a) also found that the water retaining capacity of the burnt podzolic soils was significantly higher than the unburnt soils, particularly in the upper 2 cm of the soil profile but the difference was detectable to 10 cm. This was probably also due to the presence of the ash deposits, which would have reduced the density of the larger soil pores and produced a concomitant increase in small pore density. This would have had the joint effect of reducing the rate of water percolation through the soil and increasing its water retention capacity.

A later study discovered that the water repellancy of the surface of a well drained brown pososolic soil increased slightly after burning but this declined to lower levels one year after the fire (Mallik and Rahman 1985). However, Meyles (2002) did not find any evidence of water repellancy in a small plot experimental study of the impacts of burning on the hydrological properties of a podsolic soil on Dartmoor. It was concluded that this was due to high soil moisture values and the short, low intensity controlled management burn that was carried out for the experiment. This suggests that water repellancy effects may only follow hot fires in dry conditions, but more research work is needed to confirm this.

4.1.2 pH

pH levels in the upper layers of soils have been found to increase as a result of burning. For example Allen (1964) found that peat soils at Kirby Moor had a pH of 4.4 at a depth of 1 cm, which rose to 5.2 after burning. He also reported that other workers have found that high pH values on burnt-over land are maintained for several years. Long-term serially burnt sites on peaty podzols on Tulach Hill, also tended to have higher pH levels, although few within site comparisons were found to be significant (Stevenson *et al* 1996).

4.1.3 Peat formation and carbon dynamics

An important attribute of bog habitats is the accumulation of peat, which is unconsolidated, stratified organic material largely derived from undecomposed or partially decomposed vegetation. The rate of accumulation of peat depends on the type of vegetation present and the degree of waterlogging during and after its formation. It may be expected that removal of vegetation by burning will reduce the supply of materials that may go on to form peat. This will be most significant if litter and humus layers are burnt during hotter fires, and least when cool fast moving fires are largely confined to shrub canopies. But even when moss layers remain unburnt the removal of the overlying shrub layer may be detrimental because the drier conditions will reduce waterlogging and the suitability of conditions for *Sphagnum* and other peat forming species (see 5.7)

The specific effects of fire on peat formation appear to have been little studied. However, one long-term study of grazing and burning effects on bogs , at Moorhouse NNR in the Pennines, found that peat accumulation since industrialisation (estimated to be between 1940 and 1950) was lower on plots that had been burnt every 10 years, compared with plots that had not been burnt since 1954 (citing data from Garnett *et al* 2000, Adamson 2003). Peat accumulation was lower in all parts of four experimental blocks that were grazed and burnt (with peat accumulation varying between approximately 3 – 6 cm) compared to adjacent parts that were ungrazed and unburnt (5 – 10 cm of peat accumulation) and grazed and unburnt (5 – 12 cm). Reduced rates of peat accumulation under increasing frequencies of fire were also observed by Kuhry (1994), in a study based on charcoal analysis in the boreal zone of Canada.

Peat has a very high organic content and therefore peatlands comprise a small but important global carbon store, and peatlands in the UK are a significant component of this. Carbon accumulation rates in UK bogs which are still ecologically intact has been estimated to be between 0.5 and 0.7 tonnes of carbon ha^{-1} yr^{-1}, which is equivalent to 1.8 – 2.6 tonnes of carbon dioxide (Maltby *et al* 1992). But despite this, bogs are generally considered to be neutral in terms of greenhouse gases, because the carbon accumulation in peat is offset by emissions of methane, a very powerful greenhouse gas, resulting from the anaerobic conditions. Furthermore, carbon sequestration by peatlands may have been overestimated in the past as losses of dissolved organic carbon, inorganic carbon and particulate organic carbon in fluvial discharges were not accounted for.

Few studies have examined the impacts of burning on carbon sequestration, but it has been observed that fire prevention or reduced fire frequencies tends to increase mean soil, biomass and litter carbon levels (Jones *et al* 1990, cited in Sampson and Scholes 2000). In their studies at Moorhouse, Garnett *et al* (2000) observed that on unburnt plots, grazed areas accumulated carbon slightly more quickly than ungrazed areas. However, on average the burnt plots accumulated carbon much more slowly than either (ie about 55% of the grazed and unburnt plots). Stratigraphic evidence from a Finnish mire also suggests that frequent fires reduced carbon accumulation (Pietikäinen *et al* 1999).

Garnett *et al* suggest that abandonment of burning could provide an opportunity to increase terrestrial carbon storage and mitigate against emissions of carbon dioxide elsewhere. However, they do not take into account possible increases in methane production that might occur with increased peat production, which could counteract any greenhouse gas mitigation benefits from increased carbon sequestration. Furthermore, preliminary results from current research on moorlands in Wensleydale suggests that the capacity for carbon sequestration varies with *Calluna* growth phases (Farage, in litt. 2003; see Appendix 2). Carbon accumulation rates are highest during the building and mature phases and then decline with age. Thus the impacts of burning on carbon sequestration may be more complex than realised.

Whether or not peatlands are currently carbon sinks, it is clearly important to protect the existing store of carbon in peatlands. But about 25% of UK peatlands have been disturbed by agriculture (primarily drainage) which allows re-oxidation of the carbon and release back to the atmosphere. Thus peatlands are becoming a carbon source with estimates of the UK net shift attributable to drainage being probably equivalent to 5.1-11.8 million tonnes of carbon dioxide (Maltby *et al* 1992). Furthermore, recent scientific evidence suggests that the quantity of carbon released from peatlands in fluvial processes is now increasing (Freeman *et al* 2001), as a result of rising temperatures from climate change (warmer conditions increasing microbial activity and evapotranspiration rates) and falling water tables caused by environmental change such as moorland drainage.

Oxidation of peat soils and carbon release is likely to be exacerbated by burning, as the upper layers of peat in the absence of vegetation cover will become more exposed to drying conditions. As described below, at least some erosion of peat is also likely to occur, as well as increased losses of dissolved organic carbon. Severe fires may also lead to direct combustion of the peat with obvious direct and potentially substantial direct losses of carbon stores. Severe fires often trigger long-term and severe erosion (see below), which will lead to further substantial losses of carbon.

More research is clearly needed on the extent to which burning effects current rates of peat accumulation, carbon sequestration, overall greenhouse gas emissions and losses of stored carbon. Some ongoing studies addressing these issues are described in Appendix 2.

4.1.4 Charcoal

Charcoal particles are created in fires and it has been suggested that these can have an important soil cleansing effect (MacDonald 2000). They absorb, and facilitate the break-down by soil microbes, of compounds produced by some moorland plants (such as some feather mosses, lichens and ericoids) that inhibit the growth of other plants (Zackrisson *et al* 1996, Pietikäinen *et al* 2000, Anonymous 2001c).

4.2 Nutrient dynamics

In most upland habitats, available plant nutrients decline with time because they become locked away in the woody stems of heather, and other dwarf-shrubs, and in the very slowly decaying plant litter; which in turn results in reduced food quality for herbivorous animals. Burning, therefore, has a major impact on plant nutrients in ecosystems as the combustion of the vegetation results in the partial release of nutrients. Some may become available for uptake by plants, but substantial proportions are lost from the soil-plant system.

Nutrients may be firstly lost in smoke as particulate matter and through volatilisation. Research into heathland fires indicates that nutrient losses increase with increasing fire temperatures, although the significance of this also varies according to the nutrients in question. For example, Allen (1964) found that losses of potassium and phosphorus were several times greater in severe burns, but even then the overall proportions lost were small (Table 4.1). High proportions of the carbon, nitrogen and sulphur content of the vegetation were lost in the smoke, but these losses were less effected by the fire temperature.

Table 4.1 Plant nutrient losses in smoke from burning *Calluna* (Allen 1964)

	Total % losses (by difference)*1	
	Normal burn (550-650 ^{0}C)	**Severe burn (800-825 ^{0}C)**
Weight	62.2	76.2
Potassium (K)	1.4	4.9
Calcium (Ca)	0.1	2.4
Magnesium (Mg)	0.4	2.1
Carbon (C)	60.5	67.5
Nitrogen (N)	67.8	76.1
Phosphorus (P)	0.6	3.5
Sulphur (S)	50.2	56.3

Note. * 1 Percentages of the original content in the heather, corrected by dry weight.

Similar increases in nutrient loss in relation to fire temperature have also been observed in other studies of heather burning (Kenworthy 1964, Evans and Allen 1971). Evans and Allen also found that substantial amounts of sodium, iron, zinc and copper were volatilised in their field and laboratory experiments.

There appear to be few data available on the effects of burning grasslands on nutrient losses, but one study measured nutrient losses in smoke from burning vegetation taken from calcareous grassland in the Derbyshire Dales (Lloyd 1971). This found losses of 73% nitrogen and between 25-39% phosphorus, 39-45% potassium and -1-9% calcium. Although the temperature reached by the burning material was not measured, it was noted that the method may have resulted in hotter conditions than would be expected in the field, thus possibly exaggerating the loss of some elements. The loss of more than 70% nitrogen and very little calcium is generally in agreement with other findings, such as those noted above, but the losses of phosphorus and potassium are considerably higher than reported elsewhere.

A further source of nutrient loss following burning is through the erosion of ash. As described in Section 4.3, post fire erosion can be substantial and, therefore, nutrient loses via this process may be significant. However, Kinako and Gimingham (1980) point out that such losses may only be local and that some replacement may occur from material gained from post-burn erosion elsewhere. Similarly, losses from smoke may also be partly replaced as it has been shown that some fractions of smoke condense at relatively short distances from their source fire, although it has not been possible to measure the amounts deposited (Allen 1964, Evans and Allen 1971).

But it should be remembered that any ash or soil redistribution effects will not benefit the uppermost burnt areas (as there will be no eroded ash supply to them) and may not be able to compensate for large scale fires where severe erosion may take place.

Remaining ash is incorporated into the soil and although many nutrients are readily dissolved, most are retained in the surface organic layer, if present (Allen 1964, Allen *et al* 1967). Concentrations of soil nutrients tend to be high for the first two years or so after a burn (Hansen 1969). As a result, regenerating heather shoots may contain up to twice the amount of nitrogen and phosphorus as shoots from pre-fire bushes, although this effect disappears after four to five years. Leaching can also be significant within this period if the soil has a low adsorption capacity (Robertson & Davies 1965) and losses may be further enhanced after autumn burns due to increased winter run-off (Shaw *et al* 1996).

Some studies have attempted to assess the overall impacts of burning on nutrient budgets. An early study of heathland nutrient budgets suggested that fires progressively depleted nutrient levels (Elliott 1953), but subsequent studies suggest that most elements can be made up by rainfall. Allen (1964) concluded that any loses for the soil-plant complex would be restored within a short period from rainfall inputs except on porous soils. Similarly, Robertson and Davies (1965) calculated that under normal burning conditions, as well as more severe burns where the litter is consumed, calcium and magnesium losses would be replaced by rainfall inputs within a 10-year burning cycle (Table 4.2). However, there would be insufficient levels of potassium, phosphorous and inorganic nitrogen to compensate for losses, especially under severe fire conditions.

Table 4.2 Predicted nutrient losses from burning Calluna on 10-year rotations and nutrient inputs from rainfall (based on data provided in Robertson and Davies 1965)

	Losses kg ha^{-1}		
	Normal burn (vegetation only)	**Severe burn (vegetation and litter)**	**Rainfall inputs over 10 years (kg ha^{-1})^{*2}**
Calcium	28	38	70
Potassium	43	47	30
Magnesium	12	15	40
Phosphorus (P)	7	10	negligible^{*1}
Nitrogen (NO_2 + NH_3)	119	175	90

Notes. ^{*1} The phosphorus content of rainwater is extremely low (<0.05 ppm). ^{*2} Based on gross averaged levels for 1959-61 at Aberdeen, Edinburgh and Eskdalemuir.

Kinako and Gimingham (1980) also estimated the losses of certain nutrients as a result of erosion on burnt heathlands in north-east Scotland. They calculated that after typical heathland fires soil nutrient losses would on average be replaced within one year for sodium, six years for calcium and magnesium, 15 years for potassium, 26 years for nitrogen and 75 years for phosphorus. From this, they concluded that only in the case of phosphorus was the amount substantially greater than that of potential inputs in rainfall during the inter-fire period. But this conclusion is at odds with their own calculations, which suggest that there may also be shortfalls of nitrogen and potassium if burning is carried out on typical 10-15 year rotations.

Consideration of compensation for nutrient losses through rainfall inputs should also take into account geographical and temporal variations in rainfall chemical properties. For example, oceanic areas have rainfall which has low levels of nitrogen and phosphorous and high levels of mineral salts compared to more continental areas (Coulson *et al* 1992). Atmospheric pollutant loads will also have changed since the experiments described above as a result of changes in industrial activity, advances in technology and the growth of car ownership and use. Lastly, it should be remembered that not all nutrients present in the rainfall will be retained by the soil.

The potential vegetation impacts from these apparent shortfalls in nitrogen and phosphorus from burning do not appear to have been studied. However, it has been suggested that any shortfall in nitrogen levels may be compensated for by increased microbial activity and mobilization of some of the fixed soil nitrogen (Fowells and Stephenson 1934, Tamm 1950, Allen 1964). The activity of the soil microbial population with respect to nitrogen appears to have been little studied, but Hobbs and Gimingham (1987), report that Maltby (1980) observed an increase in bacteria on peat after fire.

As pointed out by Hobbs and Gimingham (1987), the generality of the predictions made from the nutrient budget studies may be questionable as they are based on relatively cool fires of no more than 400 ^{0}C. As discussed in Section 3.2, this is unlikely to be typical of many prescribed management burns, and certainly not most wildfires.

Perhaps the most important limitation of these nutrient studies is that they infer long-term impacts of serial burning from observations of short-term effects on individual fires. Stevenson et al (1996) therefore adopted a different approach to investigating the impacts of serial burning on soil nutrient stores by studying heather stands in Scotland that differed in their long-term fire histories. This revealed that serial burning resulted in higher levels of, total potassium, magnesium, sodium and total and available manganese, while unburnt sites had significantly higher levels of organic matter, moisture, total nitrogen and available calcium. Levels of total phosphorus did not differ between burnt and unburnt stands, which contrasts with expectations based on the burning experiments reviewed above.

Overall, Stevenson et al concluded that serial burning at the study sites "increased, or at least depleted less, the mineral fertility of the soil in comparison with unburnt stands". They also noted that high levels of nitrogen deposition as a result of atmospheric pollution may have masked any effect of serial burning on nitrogen.

No attempts appear to have been made to calculate nutrient budgets with respect to grassland burning.

The overall conclusions regarding the effects of upland burning on nutrients is that in the short-term nutrient availability is increased, which stimulates vegetation growth and nutrient levels temporarily. But nutrients are undoubtedly lost from the ecosystem, and the long-term impacts of such losses from serial burning are uncertain. Most studies have only looked at single burns rather than the cumulative effects of serial burning and investigations have so far measured direct losses through volatilisation or erosion, but not both.

As noted by Coulson (1992), serial burning and its associated nutrient losses may be leading to a slow long-term decline in the nutrient status of moorland ecosystems, and therefore the productivity and quality of the vegetation for livestock, grouse and other herbivores. This may in turn have unknown knock-on impacts on other components of the foodchain and ecosystem. These could be detrimental for biodiversity if for example the declining productivity reduces food resources for certain species of conservation importance below critical levels. On the other hand Gimingham (1995) has pointed out that there has been a marked change in attitude since concerns were raised between 1963 and 1970 that declines in productivity in upland heathlands may result in declines in their conservation value. He notes that "what was not fully recognised was that it is precisely because grazing and burning tend to prevent the accumulation of nutrients in the systems that the natural trends in favour of replacement of heather by grasses, trees or bracken have been inhibited, and the continuance of a vegetation adapted to low nutrient status ensured."

Indeed, it might be concluded from this that the depletion of nutrients through burning may be particularly important under current circumstances where atmospheric pollution has significantly increased nutrient inputs from rainfall beyond natural levels (NEGTAP 2001). The consequences of such eutrophication has been well documented on many European lowland heathlands, where increased nitrogen inputs have led to the replacement of heathland vegetation by grasses and a decline in *Sphagnum* (de Smidt and van Ree 1991, Aerts and Heil 1993) There is also some evidence from *Countryside Survey 2000* (Haines-Young *et al* 2000) that, even on the UK's more isolated upland habitats, there has been an increase in plants associated with nitrogen-rich conditions, probably as a result of atmospheric pollution.

But the situation is clearly complex and variable in the uplands, and much further research is required to develop long-term nutrient budgets that can take into account all nutrient losses and gains on serially burnt areas before conclusions can be made.

4.3 Erosion

A considerable number of studies cite disturbance by fire as a possible major cause of peat erosion (eg Phillips *et al* 1981, Tallis 1987, Stevenson *et al* 1990, Rhodes and Stevenson 1997, Johns 1998). Such erosion is potentially one of the most important impacts of burning in the uplands, along with changes in vegetation. Although erosion is a natural pedogenic process particularly in areas with incomplete vegetation cover and steep slopes, it has increased significantly in the uplands as a result of deforestation, burning, intensive grazing and drainage. For example, erosion rates in Snowdonia between 1965-75 were estimated to be approximately 0.42 t ha^{-1} yr^{-1} [427 kg ha^{-1} yr^{-1}] compared with 0.05 t ha^{-1} yr^{-1} [51 kg ha^{-1} yr^{-1}] in 1800 (Dearing *et al* 1981). Research in Scotland has shown that debris-flow formation is much more frequent and more destructive than in the past (Ballantyne 1986), probably as a result of recent land-use changes, in particular burning and intensive grazing (Innes 1983). Innes cited a number of recorded debris flows and instances of severe erosion that occurred after moorland fires. Similarly, widespread sheet erosion of the blanket peats that has occurred in the southern Pennines in recent centuries may be the result of human interference with the vegetation through grazing and burning (Tallis 1964).

Increased erosion occurs after burning as a result of the loss of vegetation cover, particularly after hot fires when all plant material including surface layer mosses etc may be consumed (see Section 3.2). This loss of cover exposes the substrate to the direct effects of the sun and wind, which can lead to deflation of soil particles (ie where dry unconsolidated material is blown away by the wind). Where severe fires have removed all vegetation and most of the root mat then deflation hollows may develop, which are alternately flooded and desiccated, and these are inhospitable to recolonising pioneer vegetation. Exposure also increases the intensity of freeze/thaw cycles, which loosens the soil and makes the upper layers more susceptible to entrainment by rain and snowmelt.

A study of post-fire erosion of podzols to peaty gleys overlain with 10-40 cm of raw *Calluna* humus, at Hodge Beck in the North Yorkshire Moors, found that erosion rates were predominantly affected by vegetation cover (Imeson 1971). The rates of erosion were found to decline with plant cover (Table 4.3) and regression analysis suggested that on average accumulation rather than erosion occurred once *Calluna* height had reached about 24 cm.

Table 4.3 Soil erosion or litter accumulation beneath *Calluna* at Hodge Beck in the North Yorkshire Moors (Imeson 1971)

Condition of *Calluna* or ground surface	Mean rate of litter accumulation (+) or erosion (-) (mm yr^{-1})	Number of observations
Calluna 30-40 cm high; complete canopy	3.81	60
Calluna 20-30 cm high; complete canopy	0.25	20
Calluna 15-20 cm high; 40-100% canopy	-0.74	20
Calluna 5-15 cm high; 10-100% canopy	-6.4	20
Bare ground, surface burnt *Calluna* and raw humus	-9.5	19
Bare ground, surface of peaty or mineral sub-soil	-45.3	21

Litter accumulated beneath thick canopies of *Calluna* but on recently burnt ground with incomplete cover there was a net loss of material. Where burning only destroyed the vegetation cover and fibrous remains of burnt *Calluna* formed the ground surface, relatively slow rates of erosion were recorded. There was no evidence of surface run-off occurring at these sites and Imeson concluded that material was being removed by eluviation, wind erosion or solution. But where burning had been more severe and peaty or mineral subsoils were exposed, the rate of soil erosion was much higher, especially where the water table was high. Erosion rates on exposed soils were also found to be up to 10 times higher in the winter (September to April). This was probably due to the effects of increased running water and needle-ice development in winter. Summer and winter erosion rates were not significantly different where the ground surface was made up of burnt *Calluna* and the soil was not exposed.

In addition to accelerating the loss of surface material, Imeson (1971) also found evidence for an increase in the rate of formation and growth of shallow gully networks and seepage faces. Removal of the vegetation cover increased both surface and subsurface flows, and Imeson argued that the increase in the underground flow through fissures and channels in the humus layer would expand the gully network by undermining and collapsing surface layers.

The overall impacts of moor burning in the Hodge Beck catchment were considered by Imeson (1971) to be substantial. Citing Douglas (1969), he stated that the scale of erosion, largely as a result of moor burning, "is shown by the remarkably high sediment discharge of the river, which at 480 t yr^{-1} km^{-2} [487 tonnes yr^{-1} km^{-2}] is the highest rate of sediment discharge so far recorded for a British river".

But some caution may need to be taken in interpreting the study by Imeson as the observations of litter and soil depth changes may have been limited by the precision to which they could be measured. Also, the original PhD (Imeson 1970) indicates that the study area with short *Calluna* was regenerating on a severely burnt site from which all the peat had been removed exposing a sandy soil horizon (MacDonald pers. com., 2003). High soil erosion rates in such situations may not reflect rates where peat layers remain after properly controlled prescribed management fires. Extrapolation of the conclusions from Imeson's study to prescribed burning impacts in general also needs to take into account the extent to which the studied burns equate to those of typical management burning, but this is unclear.

A later study of two upland *Calluna* dominated heaths in north-east Scotland, by Kinako and Gimingham (1980), found that the surface level of the peaty podsol soils was reduced by 0.27-0.55 cm following burns (although there was considerable variation across the 15 'erosion stakes' in each burnt site). Assuming that this reduction in level was caused by a loss of particles from the surface, these results indicate that soil losses amounted to 1,270-3,850 kg ha^{-1}. However, the soil losses took place mainly during the first eight months and declined as the vegetation recovered. By 15-20 months after burning, vegetation recovery had completely arrested the process. They concluded that under good moor management, with controlled burns of appropriate vegetation stands, vegetation recovery is rapid, and therefore serious soil loss and habitat depletion is unlikely. But the study did not measure erosion / accretion rates on unburnt control areas or on the burnt study areas beyond 24 months after burning. Thus it is not possible to say how erosion rates compared with background levels on vegetated land, or how long it would take for accretion (assuming this occurs on vegetated areas) to make up for the post-fire soil losses. Nor is it possible to quantify long-term soil erosion impacts from cyclical burning.

Kinako and Gimigham also studied the effects of slope on soil erosion rates with the expectation that losses would increase with increases in slope. But such a relationship was only found at one of their two study sites. They also reported that Imeson (1971) failed to find a relationship between slope and erosion. Sampson (1944, cited in Kinako and Gimingham 1980) concluded from studies of chaparral burning in California, that while in some areas soil erosion was proportional to angle of slope, in others there was no such relationship. Nevertheless, Kinako and Gimingham conclude that erosion losses may be great where the upper reaches of steep slopes are burnt (as these are inherently more unstable) and where an excessively large area is burnt at one time. Increased erosion impacts are likely to result from larger fires because they tend to be hotter and more intense (see Section 3.2), which increases vegetation loss and soil damage. Also, the larger the area from which protective vegetation is removed, the more likely it is that slope wash will build up sufficient momentum to remove soil particles (Anonymous 2001c).

A study of the erosion of dissected blanket peat in mid-Wales, where burning is not considered an important factor (Francis 1990), found that lowering of the peat surface in summer was mostly due to wastage and shrinkage of the peat, and that the major removal of peat as particulate matter took place in autumn and winter, following loosening by ice, especially on north- and east-facing surfaces. Francis estimated that 23% of the total suspended organic sediment was removed by flood events that lasted 2% of the time, highlighting the significance of changes in the quantity and timing of run-off from bare or burnt surfaces. While a flashier run-off regime might not add to the total output of organic sediment from a peat catchment, because the supply of transportable granular material is rapidly exhausted with the onset of winter, it is likely to lead to high peak discharges coinciding with high concentrations of particulate organic matter and colour.

Thus, although there is some uncertainty over the actual quantities of soil lost following management burns, the results emphasise the high risks of erosion following ‘clean burns’, where all vegetation and litter is removed (often in the mistaken belief that this encourages revegetation). There is also likely to be a high risk of erosion after extensive agricultural burns and other burns that get out of control and cover large areas. These may share some characteristics of wildfires, which are often similarly large and intense. They are more likely to cause physical changes to the soil surface and loss of the root mat, and may cause deep and extensive combustion of peat and humus layers. Evidence to support this comes from the impacts of severe fires at Rosedale Moor, where subsequent heavy rainfall caused considerable erosion of the burnt substrate as it lacked its previous vegetation cover and absorbent peat surface (Maltby *et al* 1990). The consolidated light powdery ash was particularly quickly eroded and gullies rapidly developed. In the most severely burnt areas the mean depth of ash was reduced after just four weeks of heavy rainfall by more than half (although this would have included some compaction). At the same site, Arnett (1980) observed a two-fold increase in sediment loading during peak stream discharges after the fire compared with previous records. He also reported that under storm conditions soil erosion on bare surfaces exceeded that under mature heather stands by at least 20 times.

Maltby et al (1990) observed that, in addition to erosion through run-off, soil loss occurred as a result of deflation, particularly during dry spells. This created shallow deflation hollows and led to the build up of ash and fine peat granules against stones, burnt vegetation and other obstructions. As observed by Imeson (1971), frost action also clearly played an important role. Needle-ice broke up the peat surface, resulting in a greater instability of exposed surfaces during winter and expansion of cracks and cavities in the peat. Most importantly the study concluded that, in the absence of vegetation cover, it can be predicted that further wetting-drying, heating-cooling and freeze-thaw cycles will lead to continued peat fragmentation and granulation and the progressive disintegration of the peat mass until the entire organic layer is lost.

In places the Pennine blanket peats are intensively dissected by gullies and a significant proportion of the ground surface is bare peat or exposed regolith. Other nearby areas have escaped dissection, and retain an essentially complete vegetation cover. Burt and Gardiner (1982) established an intensively instrumented catchment study of the generation of run-off and sediment from eroded and uneroded areas, and found that the much greater supply of granular materials from the eroded area led to consistently higher yields of suspended sediment, with concentrations closely related to the flow. The supply of sediment from uneroded areas was soon exhausted, although large amounts of run-off were produced from both areas. Vegetation was able to filter out entrained particles from the flow and protect the ground surface against erosion.

Past severe fires may be a widespread cause of the initiation of peat erosion. For example, the onset of widespread erosion at Holme Moss, in the southern Pennines, appears to have been associated with at least one severe fire in the 18th century, though probably exacerbated by a freak cloudburst (Tallis 1987). Similarly, at least some of the expansion of areas of bare peat in the Peak District appears to be the product of past fires linked to climatic factors (Radley 1965, Anderson 1986, 1997). Mackay and Tallis (1996) attributed the onset of summit-type erosion in the Forest of Bowland to a combination of unusual factors: a period of below-average rainfall in the region in the early 1900s, resulting in lowered water tables in the peat; and a decline in standards of moorland management because of a shortage of gamekeepers after World War I. This may have reduced the frequency of burning leading to an increased risk of wildfires. The final event triggering the erosion process may then have been a catastrophic wildfire, probably during the drought year of 1921, which may well have been accidental in origin.

Erosion impacts can have obvious knock-on effects on ecosystems. After severe fires large areas of rock, regolith and peat may be exposed, and as these are hostile to plant growth they may remain unvegetated for many years (eg Tallis 1973, Maltby *et al* 1990). In particular, there are likely to be significant downstream impacts, possibly on a catchment scale, as increased sediment loads and dissolved erosion products will affect the physical and chemical environment of rivers and lakes. Increased sedimentation is also likely to have been exacerbated in many upland areas as a result of moorland drainage (although in some areas this is being reversed), increased grazing pressures and afforestation, all of which also tend to increase sediment run-off.

Impacts on river and lake vegetation may include blanketing of submerged plants by sediment, difficulties with rooting in deeper, softer substrates that are susceptible to scouring in spate conditions, and reduced photosynthesis through increased turbidity (Backshall *et al* 2001). This may change the aquatic community from a mixed community with submerged and floating plants, to one that is dominated by floating-leaved species, and perhaps eventually emergent plants and no aquatic macrophytes.

Invertebrates and fish are also likely to be affected by high sedimentation rates, as the substratum will change from rocks and gravel etc to soft sediments, which will favour a different faunal community and will be unsuitable as spawning grounds for salmonid fishes.

4.4 Hydrology

Point measurements, eg the determination of soil hydraulic properties described in Section 4.1.1, do little to answer the question of the significance of upland burning on the catchment scale, ie its importance or otherwise to downstream ecology, water supplies or river management. Furthermore, relatively few studies appear to have been carried out on the hydrological impacts of burning at the plot or catchment scale, and those that have been carried out in British uplands have produced partially conflicting results. The problems of experimentation on the small catchment scale are well-known: for catchments of less than about 1 km^2, it may be difficult to define the catchment boundary, and to find a suitable location to measure streamflow. The smaller the catchment, the less reliable will be the measurement of both inflow (from precipitation) and outflow (which may comprise both channel flow and diffuse throughflow or groundwater seepage). The typical scale of controlled heather burning is about 0.5 ha (Phillips 1981), far too small for a useful research catchment. Studies could be conducted on larger catchments with multiple fires, but the fires will inevitably differ in their characteristics making interpretation of the hydrological impacts more difficult. Valid comparisons against unburnt catchments (ie experimental controls) will also be difficult.

Accidental fires by their nature are unpredictable in timing and location, and unsuitable subjects for catchment experimentation. Not surprisingly, there appear to be no scientific results relating to the downstream hydrological impact of wildfires.

As mentioned in Section 4.1, burning has been shown to reduce the infiltration of water into the soil, which may thus increase surface flows during rainfall. However, in an experimental study on Dartmoor, no increased surface flows were observed as a result of simulated heavy rainfall following a controlled management burn, at least at the plot scale (Meyles 2002). Neither were any differences in soil moisture levels detected by Meyles under wet conditions between burnt and unburnt areas, for up to two months after burning.

Severely burnt soils would be expected to generate an increased proportion of rapid run-off, through increased water retention in the soils and the absence of evapotranspiration and rainfall interception by vegetation (Mallik *et al* 1984a).

One of the few catchment studies on blanket peat was carried out in the 1950s at Moor House in the Pennines, the results of which suggested that severe burning of the peat surface reduced the water storage capacity of the soil and lowered dry weather flows, but increased peak flows in drainage ditches (Conway and Millar 1960). By analysing a longer sequence of Conway and Millar's data, Robinson (1985) confirmed their conclusions. Though moderate burning, without total destruction of the mire community or the heather, tended to produce a flashier run-off hydrograph, the effect was very much less than that of drainage by gripping.

5. Impacts on habitats, vegetation communities and plant species

Most studies of the impacts of burning on plants have focussed on plant community responses, or impacts on their dominant and characteristic species. Impacts on specific species are therefore discussed in this chapter according to generic issues and in relation to the habitats that they are associated with. However, a summary of impacts on key plants species and species groups is presented in Table 5.2 at the end of this chapter. Discussions of issues concerning particular species can also be traced by reference to the species index provided at the end of this report.

5.1 General impacts

Most upland habitats, with the exception of those that are waterlogged or above the potential tree-line, are plagio-climax habitats, ie are formed and maintained by human disturbances that interrupt the natural successional processes. Furthermore, many of the characteristic species of these upland habitats, such as *Calluna*, and other dwarf shrubs are plants of seral vegetation (ie successional stages) rather than climax communities, and their long-term presence is maintained by grazing and fire (Gimingham 1971, Gimingham 1972, 1995). Thus the most usual and important impact of burning upland vegetation is the maintenance of open shrub or grass dominated habitats as opposed to the development of scrub and woodland. Though in some cases where old heath stands are burnt this may actually encourage scrub invasion, as *Calluna* vegetative regeneration is poor from burnt old stands (see Section 5.2 below). The burnt ground also creates a good seedbed for trees such as *Betula* and *Salix*; so seedling establishment of these species can be profuse if a seed source is present. If fire does not return for 30 or 40 years then the trees will be to tall to be killed by fires and the field layer will have changed character sufficiently to reduce the likelihood of carrying a fire.

Prediction of the exact vegetation communities that would develop in the absence of regular burning is difficult because the tendency for, speed and outcome of secondary succession processes varies greatly from place to place. Succession appears to primarily depend on four factors: the availability and diversity of seeds, fruits and other propagules of potentially competitive species, soil fertility, existing vegetation structure (as some species can form dense stands that resist invasion without disturbance) and the intensity of grazing and burning management (Miles 1988). Thus there tend to be multiple pathways of potential change, rather than fixed directions of change. For example, Figure 5.1 summarizes observed successional changes between eight common types of semi-natural vegetation on well-drained, acid mineral soils, according to three broad levels of grazing intensity and burning frequency. Unfortunately the author does not define “frequent” or “occasional”, but the general implications of burning are apparent.

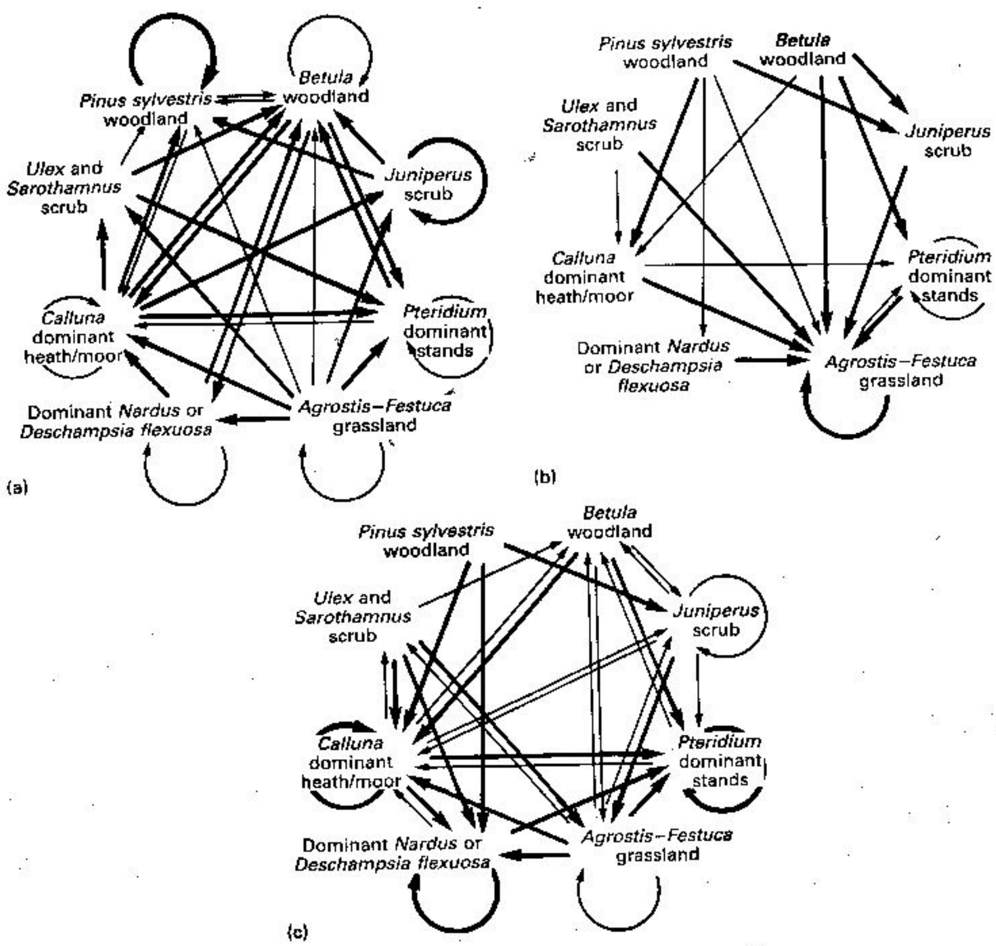

Figure 5.1 Successional transitions in the uplands (particularly north-east Scotland) between eight vegetation types according to three combinations of grazing intensity and burning frequency (reproduced from Miles 1985 courtesy of the British Society of Soil Science & Blackwell Publishing Ltd).

(a) low grazing pressure (<1 sheep equivalent ha^{-1} yr^{-1}) and no burning; (b) high grazing pressure (2-3 sheep equivalents ha^{-1} yr^{-1}) and frequent burning; (c) intermediate levels of grazing (1-2 sheep equivalents ha^{-1} yr^{-1}) and occasional burning. Broad arrows represent common transitions, thin arrows apparently less frequent transitions, and curved arrows self-replacement. The vegetation types are arranged so that types tending to podzolize and/or acidify soils are on the left, and types with contrasting effects on the soil are on the right.

With low grazing pressures and no burning (Figure 5.1a) several habitat types may be able to co-exist as a shifting mosaic of stands as they establish, die-off and become replaced. An illustration of this, is provided by Miles (1981), in which a generalized sequence of establishment, senescence and re-establishment of birch, grass and heather is described, with associated soil changes, in birch woods in Scotland. There is also potential coexistence of multiple habitats under the scenario described for Figure 5.1c, but this lacks the woodland component and the principal transitions are to heather moorland, rough grassland and *Pteridium* stands. With heavy grazing and frequent burning, the tendency is to develop *Agrostis-Festuca* grassland.

Miles (1988) suggests that it is the availability of alternative dominant species that is the main reason why any particular transition occurs. But in many uplands potential successional woodland and fire sensitive species are lacking, because these have been exterminated by deforestation and subsequent burning. Thus there tends to be a greater variation in successional vegetation at moorland edges, whilst over large areas of interior moorland succession is inhibited because potential successional species are absent in the vegetation, the seed bank and the dispersal ‘rain’ of propagules.

Prediction of the more immediate outcome of vegetation re-establishment following a fire is similarly complex as it depends on a number of interrelated factors (Figure 5.2). Nevertheless, three key factors appear to largely determine the impact of a fire on post-fire vegetation re-establishment:

- pPre-fire vegetation composition and age;
- fire intensity; and

post-fire management, especially relating to grazing.

Vegetation development after a fire varies considerably in terms of its rate and its species composition. But in most circumstances following a controlled management burn this variation is predominantly related to stand age and its effect on the community composition before the fire (Hobbs *et al* 1984, Hobbs and Gimingham 1984a, 1987). This is primarily because the pre-fire vegetation will be the source of the regeneration, which following typical managed fire conditions will be primarily vegetative.

Burnt areas that have not been affected by fires in the recent past may contain species that are fire sensitive, ie they have no means of surviving the fire or rapidly regenerating afterwards. But in most moorland areas in Britain the vascular plants present will have some form of fire survival mechanism (Hobbs *et al* 1984). For example, species such as *Vaccinium myrtillus* have below-ground rhizomes which are unlikely to experience high temperatures because soil and moist litter provides good insulation, particularly when damp (see Section 3.2). Above ground buds may also be protected by sheathing leaf bases (such as in *Molinia*) or woody bark material and litter (as in *Calluna* and some other ericaceous shrubs). Table 5.2 provides further information on fire survival mechanisms in some key upland plant species.

Most species will therefore be able to survive or regenerate from typical management fires. However, MacDonald (2000) notes that there are a few species with shallow root systems, no rhizomes, creeping or low growing surface stems, and little capacity for resprouting from stem bases that may be killed even by quite low intensity fires. Such species include *Empetrum nigrum, Linnaea borealis*, *Juniperus communis*, *Veronica officinalis* and some epiphytes such as the lichen *Hypogymnia physodes*.

Vegetation re-establishment after most typical management fires is by vegetative means (as seedlings are out-competed by the vegetative growth) and usually goes through a series of phases depending on the dominance, diversity and condition of the various species in the pre-burn community. In heathlands, such phases have been explained in terms of the ‘regenerative strategies’ of the species present, which reflect their life form and speed of shoot development (Mallik and Gimingham 1983, Reader *et al* 1983).

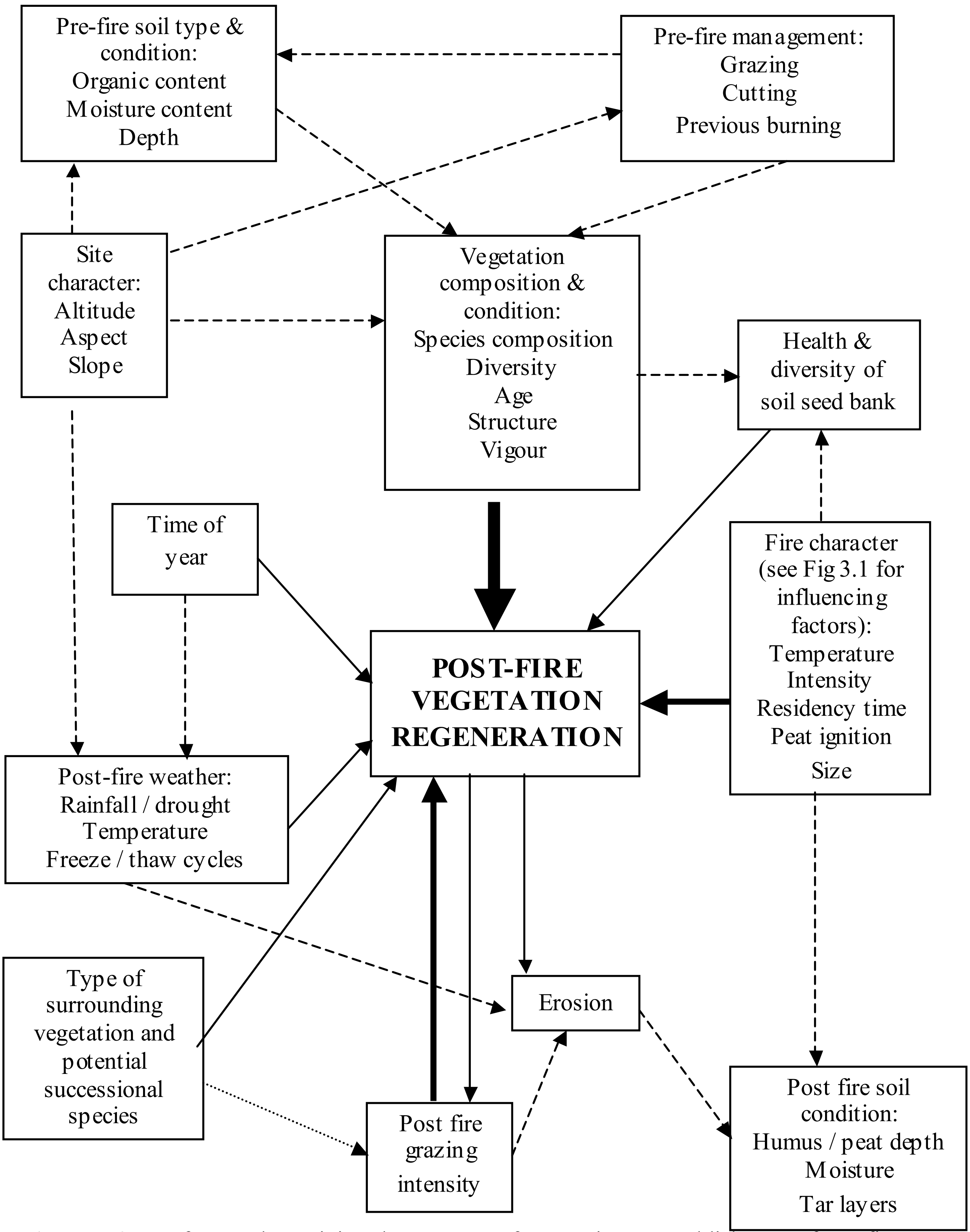

Figure 5.2 Key factors determining the outcome of vegetation re-establishment after a fire and their interrelationships.

Line widths are indicative of the approximate importance of the factor. Dashed lines indicate indirect influences. Factors influencing fire character are excluded (to avoid over complication) and these are summarised in Figure 3.1.

Regeneration from seed sources becomes more important when old stands are burnt, as the capacity for vegetative regeneration declines with age in many species (Miller and Miles 1970, Hobbs and Gimingham 1984a, Berdowski and Siepel 1988, Gardner *et al* 1993). Fires with high ground intensities may also damage rhizomes and basal shoots thereby reducing the potential for vegetative regeneration (see below). Many species form seedbanks and those of *Calluna* are very long-lived, but seed production and viability typically declines with stand age. This was demonstrated by Mallik *et al* (1984b) in north-east Scotland, who found a dramatic reduction in germinable seeds of nearly all species from stands older than about 16 years, with the exception of seeds of *Calluna* and *Carex pilulifera*. Species germinating from the seed bank declined from an average of about 7 to about 2.5 in stands ranging from 5 to 35 years since burning. But some species then showed an increase in germination in *Calluna* stands that had reached the degenerate stage (at about 30-40 years), most likely as a consequence of increased seed production from a few surviving individuals. Thus the frequency of serial burning will be particularly important as the age of the stand will in most circumstances depend on when it was last burnt.

Some species have very short-lived seed banks or do not form any at all (Table 5.1). These species are therefore unlikely to be able to recover following very hot fires or fires in old stands.

Table 5.1 Moorland species with no seed bank or a short lived one according to MacDonald (2000)

Based on (Hill and Stevens 1981, Hobbs *et al* 1984, Mallik *et al* 1984b, Granström 1987, 1988, Hester *et al* 1991, Granström and Schimmel 1993). Species in brackets have seed banks which show substantial declines over a decade. Species followed by a question mark are those for which evidence is sparse, uncertain or conflicting.

Achillea millefolium	*Juniperus communis*	*Sieglingia decumbens ?*
Anemone nemorosa	*Lotus corniculatus ?*	*Stellaria media ?*
Anthoxanthum odoratum	*Luzula campestris*	*Trichophorum cespitosum ?*
Arctostaphylos uva-ursi	*Luzula spp.*	*Trientalis europaea*
Deschampsia flexuosa	*Plantago lanceolata*	*Vaccinium myrtillus ?*
(Erica cinerea)	*Poa pratensis ?*	*Vaccinium visit-idaea ?*
Hieracium pilosella	*Potentilla erecta*	*Veronica chamaedrys ?*
Holcus lanatus	*Pyrola media ?*	*Viola riviniana ?*
(Hypericum pulchrum)	*Rumex acetosa ?*	

Another key determinant of post-fire succession is the character of the fire (Webb 1986), in particular its temperature and ground surface intensity, which are primarily determined by the pre-fire composition and structure of the stand (see Section 3.2). Hot fires with a high ground surface intensity may kill off the shoot bases of species such as *Calluna* (especially on dry substrates, where insulation is poor), and thus re-establishment of such species will rely more on seed. If the ground surface intensity is very high, then seedbanks may be killed, or even consumed if a smouldering fire ignites the humus or peat layers. If no viable seedbank is present because of the effects of the immediate or previous fires then re-establishment will be reliant on seed rain or slower colonisation from the surrounding vegetation. In such cases recolonisation may be very slow and initially restricted to bryophytes. There may also be invasion by scrub or *Pteridium* if these are present in the vicinity.

Where fires have been very severe, then loss of the overlying vegetation, litter, humus and peat layers if present may lead to unstable and hostile conditions for plant seedlings. As described in Section 4.1.1 highly fluctuating temperatures, drying conditions and tarry crusts may occur on the bare soil surface, and seedling establishment may be low under such conditions (Mallik and Gimingham 1985, Mallik *et al* 1988). Furthermore, the lack of vegetative cover encourages erosion (see 4.3) and further soil loss, and such unstable conditions will further inhibit vegetation re-establishment. As noted before, many upland areas that have suffered severe fires have remained unvegetated for many years after (Tallis 1973, Maltby *et al* 1990).

A third key factor affecting post-fire vegetation re-establishment is the effect of post-fire management, in particular grazing. Many areas of the uplands are currently affected by high levels of grazing pressure (Brown *et al* 2001), which may be exacerbated by burning. Burning removes old, unproductive, woody plant material, allowing the re-growth of young, vigorous shoots with a high nutrient value (Gimingham 1972) and therefore livestock tend to concentrate in recently burnt areas. This combined with currently high stocking rates in many areas can result in high grazing and browsing pressures on re-establishing vegetation (Grant and Hunter 1968). This may deflect post burn successions away from re-establishment of the pre-burn vegetation.

Grazing may also slow down vegetation re-establishment particularly after severe fires. This was demonstrated by the creation of an exclosure on a severely fire damaged area of the North York Moors, which reduced grazing pressure and led to faster recolonisation by moorland species (North York Moors National Park 1991).

With the exception of a relatively few small experimental exclosures virtually all upland areas are subject to grazing by livestock, and therefore burning impacts in the absence of grazing are not well understood. Indeed, it is somewhat inappropriate to review and discuss impacts of fire on vegetation in isolation from grazing impacts, as burning and grazing are inextricably linked. Key interactions between fire and grazing that affect the impacts of burning are therefore also discussed below. But it has not been possible to carry out a detailed assessment of the interactions between grazing and burning in this review. Similarly, other factors that affect vegetation in the uplands may also influence post-fire vegetation re-establishment, such as drainage, nitrogen deposition and other forms of atmospheric pollution, but interactions between these do not appear to have been investigated.

An important conclusion to be drawn from the examination of factors influencing post-fire vegetation re-establishment is that there is no simple predictable post-fire succession. Instead there are multiple pathways where the direction of development depends on a number of factors (Hobbs *et al* 1983).

Further examinations of the impacts of fire on vegetation are presented below according to habitat type because the pre-fire composition of vegetation is the predominant influencing factor. These sections are divided according to UK BAP Broad habitat types and only those upland habitats that are regularly burnt for management purposes are considered. The impacts of burning *Juniperus* scrub is discussed in the sections on dwarf-shrub heath, as *Juniperus* is only normally burnt when it is a scattered component of this habitat, not when it forms woodland stands.

The effects of other influencing factors on fire impacts, such as fire intensity, size and frequency, burning method and post-fire management, are discussed in relation to effects on each broad habitat type and summarised in the conclusions chapter (Section 7.3).

5.2 Dwarf-shrub heaths

5.2.1 Dry heaths

Much of the research into the impacts of fire on upland habitats has focussed on dry heaths and therefore this section reviews this work in some detail. Many of the results, however, are applicable to other heaths and upland habitats, particularly those with *Calluna* as a significant component. Dry heaths also warrant special attention as they are listed under Annex 1 of the Habitats Directive and the UK resource is of particular importance (Table 1.1). They also fall within the Upland Heathland UK BAP Priority Habitat.

Pre-fire vegetation composition and age

Dry heaths are typically dominated by ericaceous dwarf-shrubs, and as a result they tend to burn readily and produce relatively hot temperatures (see Section 3.2). *Calluna* is the dominant component of most dry heaths, and the impact of fire is largely determined by the condition of *Calluna* in the pre-fire community. This is because the recovery ability of *Calluna* varies according to its age and growth phase, as this largely determines whether re-generation is mostly vegetative or from seed.

Calluna is well adapted to surviving fire as it can re-establish vegetatively from buried stools or basal shoots, and rapidly becomes reproductively mature (Hobbs *et al* 1984). If stands up to the mid-building phase (ie where stems are up to 4-5 mm in diameter at the base) are carefully burnt then *Calluna* normally regenerates rapidly, virtually by-passing the pioneer phase with re-establishment of a young building stand in 1-2 growing seasons. However, *Calluna* regenerates more slowly after burning older stands (Miller and Miles 1970, Hobbs and Gimingham 1984a, Berdowski and Siepel 1988, Gardner *et al* 1993). This is because older *Calluna* develops secondary thickening at the stem base which inhibits resprouting following a fire. By the mature phase, after about 15 – 20 years most stems will have lost the ability to resprout. For example, a study by Berdowski and Siepel (1988) found that 90% of stems resprouted in stands less than 5 years old, but this declined to 10% in stands that were 20 years old. In degenerative growth stages there are also fewer stems and therefore fewer sites for potential regeneration (Hobbs and Gimingham 1987).

Similar declines in regeneration capacity have been found with *Erica cinerea* (Hobbs and Gimingham 1984a).

Calluna produces abundant seed throughout its life and there is usually a large seed bank present in the soil (Mallik *et al* 1984b, Mallik *et al* 1988, Cummins and Miller 2002). Therefore where there is little vegetative regeneration then recovery can be by seed germination. Where both occur then vegetative regrowth usually overgrows the seedlings. Germination may take place in the same season as the fire or may take longer (Hobbs and Gimingham 1984a). There is some evidence that burning stimulates seed germination in *Calluna* (Gardner *et al* 1993), although this was not supported by results from Gonzalez-Rabanal (1995). If burning does stimulate seed germination, this may be due to the effects of smoke rather than heat, as smoke treatments have been used to enhance *Calluna* germination in heather restoration work (MacDonald pers comm 2003).

Seed germination is typically patchy as it depends upon the abundance and viability of seed in the litter layer (Legg *et al* 1992). The density of seedlings in a species-rich Scottish heath was found to be highest in stands which were youngest before a fire, declined markedly in the building phase and mature stands (14-25 years), and increased again in stands which were oldest when burnt (Hobbs *et al* 1984). Seed germination is also often uneven because the occurrence of good seedbed conditions is typically patchy as fires seldom burn uniformly.

Fires in young (and frequently burnt) stands tend to produce the best seedbeds, with clean reasonably consolidated humus or mineral soil surfaces which do not readily dry out. This is because the vegetative material in young stands is finely divided and therefore all tends to be potential fuel. It also burns closer to the ground and therefore creates a fire with higher ground intensity. Older stands may leave patches of loose litter and moss mats, which provide poor seedbed conditions as they dry out too quickly, and lichen mats which may have allelopathic effects.

Large amounts of unburnt litter or debris may also remain on the soil surface following low intensity burns, and this may significantly retard regeneration from seed (Coulson *et al* 1992). This is why prescribed fires often aim for a 'clean burn', even though in many cases this is not necessary (and may be detrimental) as there may be good potential for vegetative regeneration, which is unaffected by debris. But hot burns (eg of older tall mature stands) may also reduce germination as seedling establishment can be inhibited if the soil organic matter is charred during an intense burn (Mallick *et al* 1988). A hard crust may form on the peat surface after hot burns which not only inhibits seedling regeneration but also encourages the growth of mats of algae, lichens and pleurocarpous mosses which in turn further inhibit growth (Mallik *et al* 1984a, Legg *et al* 1992).

Following germination, seedling survival and establishment is affected by a number of other factors, in particular the soil micro-climate created after a burn (Mallick *et al* 1988). This is because heather seeds are very small (c. 0.7 mm x 0.5 mm) and consequently heather seedlings are very small and have a small root system. It takes them several years to attain any significant size, and they are thus vulnerable over this period to freeze-thaw effects and even quite ephemeral drying of the seedbed. Considerable mortality of ericaceous seedlings has been observed as a result of summer drought (Mallik and Gimingham 1985).

Recovery by seed germination is very slow compared to vegetative regrowth (Hobbs and Gimingham 1984a) and this allows greater opportunities both for erosion to occur (see Section 4.3) and for colonisation by other plant species. Bare ground after a fire is normally initially colonised by algae, acrocarpous mosses, typically including *Ceratodon purpureus*, *Polytrichum juniperinum* and *P. piliferum*, and lichens, initially *Lecidea granulosa* and *L. uliginosa*, followed by a variety of *Cladonia* species (Hobbs and Gimingham 1987). Herbs and grasses (such as *Deschampsia flexuosa*) may also establish shortly after the fire, but these are normally outcompeted quickly if vigorous vegetative regrowth of *Calluna* and other ericaceous species occurs. But where recovery is via seedling germination then the early lichen and grass dominant phase may be prolonged and species other than *Calluna* may attain long-term dominance. For example, as illustrated in Figure 5.3, Hobbs and Gimingham (1984a) found that in species-poor heath in Scotland, the speed of recovery of *Calluna* varied considerably with stand age. Furthermore, where *Calluna* regrowth was largely from seed and very slow (stand M1) *Vaccinium myrtillus* was able to increase and prevent *Calluna* seedling establishment.

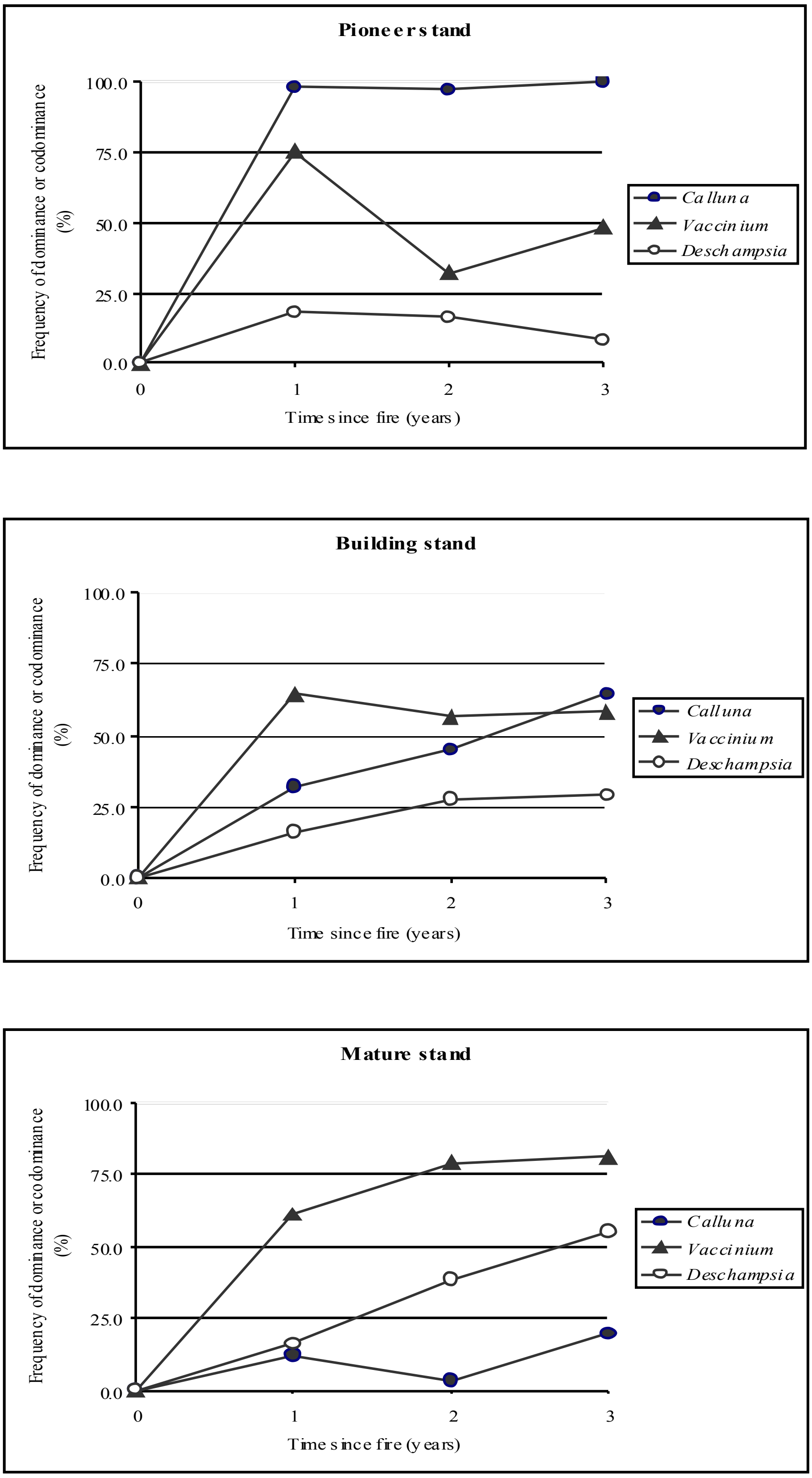

Figure 5.3 Vegetation recovery in species-poor heath of different pre-fire ages in Scotland (redrawn from Hobbs and Gimingham 1984a, courtesy of Blackwell Publishing Ltd.)

Stand ages before fire are: Pioneer = 6 years; Building = 10 years; Mature = 15 years. Percentage dominance or co-dominance by *Calluna*, *Deschampsia flexuosa* and *Vaccinium myrtillus* in 128 subplots (each 10 x 10 cm) during 1978-80.

Stand age is also important in determining the response of other species to fire. In heaths rich in herb and grass species, the number of species present in post-fire stands declines with the age of the pre-fire stand. The average number of vascular species in stands one year after a fire in species-rich heathland in Scotland varied between about 10 – 13 for burnt stands that were 5 – 15 years old, but dropped substantially to four to seven for older stands, up to at least 40 years age (Hobbs and Gimingham 1984a).

The ability of a species to recover from burning depends largely on the timing of the fire in relation to its key life history events, in particular its attainment of reproductive maturity and its capacity for vegetative regeneration and seed production. For example, many species lose the ability to regenerate vegetatively as their numbers decline with the closure of the *Calluna* canopy. Germinating seeds may also fail to establish once canopy closure has occurred (de Hulla and Gimingham 1984). The period of viability of seeds in soil seedbanks is very important as this is relatively short for some species (Mallik *et al* 1984b). For such species, without further seed supplies, there will be a time after which post-fire regeneration from the soil seedbank will not occur.

The relationship between the timing of burning in relation to the development of the community and these critical life history events has been examined by Hobbs et al (1984) for species-rich heathland in Scotland. Figure 5.4 illustrates the timing of these key life history events, in particular the availability of propagules and regeneration sites and local extinctions of species in relation to the classic *Calluna* development cycle. This in turn represents the potential for regrowth by each group at each stage of the cycle. Two lines are shown for *Calluna* to represent its persistence through vegetative regeneration (V) or seed germination (S). Thus by drawing a vertical line through the diagram it is possible to predict the species that are capable of regrowth following a fire at each stage of the cycle.

	Phase of *Calluna*				
	Initial	**Pioneer**	**Building**	**Mature**	**Degenerate**
Calluna vulgaris S	op	m			
Calluna vulgaris V	op	m		e	
Erica cinerea	op m				
Arctostaphylos uva-ursi	op		m		
Grasses	op m		l	e	p
Forbs	op m		l	e	p
Carex pilulifera	op m			l	
'Pioneer' mosses	opm		l	e	
Pleurocarpus mosses	o	p?	m		
Cladonia species	o pm			l e	
Cladonia portentosa	o	p	m		
Vaccinium spp	opm				
Pteridium aquilinum	opm				

Figure 5.4 The timing of critical life history events for major species present in species-rich heath in Scotland (redrawn from Hobbs *et al* 1984, courtesy of Blackwell Publishing Ltd)

Symbols: (o) time of disturbance; (p) time at which propagules are available on site; (m) time at which reproductive maturity is reached; (l) local loss from the community; (e) local extinction from the community. *Calluna* V represents persistence through vegetative regeneration; *Calluna* S represents persistence through seed germination.

From Figure 5.4 it can be seen that a fire in the early stages of the *Calluna* cycle will result in rapid regrowth of most species since many of them, including grasses and forbs, are present in abundance. Such species can grow vegetatively, but their seed is also present in the soil. As the cycle advances then regeneration will still be diverse, but some grasses and forbs may be absent from the community. The overall result of burning old growth stands is extremely slow regrowth (as most is by seed), reduction in species diversity (as some species are absent and the seedbanks of some of these may no longer be viable) and persistence of bare ground for many years after the fire (Hobbs and Gimingham 1984a). As noted above, burning old stands may allow spread and dominance of rhizomatous species such as *Vaccinium myrtillus*, *V. vitis-idaea* and *Pteridium aquilinum*, if they are present.

Hobbs and Gimingham (1984a) found that the effects of pre-fire species composition and stand age were still apparent in post-fire communities eight years after a fire. They concluded that post-fire development in any particular stand is largely determined by its initial post-fire floristic composition whose constituents are in turn determined by the regenerative capacity of the stand (ie the availability of regeneration sites and viability of the soil seedbank). They also suggested that there is a simple post-fire vegetation establishment hierarchy which depends on whether or not *Calluna* regenerates vegetatively. If vegetative re-generation occurs then *Calluna* will regain dominance even if aggressive rhizomatous species are present, whereas, if *Calluna* regeneration is only from seed, it is slower to develop cover, which may allow a rhizomatous species, if present, to become dominant.

Attempts have been made to produce predictive models of post-fire development of heath vegetation on the basis of pre-fire floristic composition. Hobbs (1983) produced Markov models using data from a variety of heath types, but found that satisfactory predictions could be obtained only for the simpler heath systems. However, Hobbs and Legg (1984), developed an earlier model (Legg 1978, 1980) which supported the hypothesis that stand age before burning influences the post-fire development through the process of colonisation of bare ground. The model also suggested that post-fire development after severe fires may initially depend on the formation of moss cover, although they concluded at the time that it required further study.

The predicted impacts of burning on post-fire vegetation development also appear to be supported by an examination of vegetation in long-term serially burnt heather moorland (Stevenson *et al* 1996). This study examined five sites where stands had been identifiably burnt two or three times since the 1940's, and five unburnt stands, which did not appear to have been burnt for 75 -190 years before the study. The study found that serially burnt stands were much more heterogeneous than the unburnt stands and generally had a higher species-richness. Unburnt stands were more uniform, being dominated by dwarf shrubs, particularly *Calluna*, and bryophytes (as would be predicted in Figure 5.4). But in mature stands species-richness was lower in stands that had a history of serial burning ie there were indications that a least some species were either sensitive to fire or were very slow colonisers ('old growth' species). An important point to note from this is that it is worthwhile distinguishing between species that are absent because they <u>are</u> fire sensitive and species that <u>appear</u> to be sensitive to fires (or post-fire conditions) because they are slow colonisers. Fire sensitive species will not be affected by burning patterns but slow colonisers will. Thus, burning at different spatial scales may lead to very different colonisations rates depending on the closeness of potential colonists and therefore different vegetation communities.

Stand age is normally determined by fire history, unless the stand has been cut or regenerated in some other way. Therefore the frequency of prescribed burning is one of the principle management factors that determines burning impacts. Frequent burning in the pioneer phase will favour more rapidly developing and seeding species than *Calluna*. Burning of *Calluna* dominated stands on mineral soils at three- to six-year intervals shifts dominance towards grasses, especially *Deschampsia flexuosa* on well drained soils and *Molinia* on poorly drained soils (see later Section 5.4 on acid grassland), whilst burning every six to ten years favours *Erica cinerea* and *E. tetralix* (Miles 1988). Frequent burning may also allow rapid invasion of *Pteridium* if rhizomes are present in the soil.

As described above different species tend to be present at different stages of the *Calluna* cycle (Figure 5.4). Thus as noted by Stevenson *et al* (1996) species diversity can be maximised at the landscape scale by maintaining some stands under burning management with burns carried out shortly before the mature growth phase, and some stands with no burning at all.

Fire character

The character of a fire in terms of its temperature and ground surface intensity also has a major influence on the impacts of the fire. Indeed the predicted outcome of post-fire vegetation development discussed above only applies to typical well controlled prescribed management fires and not severe fires. Exceptionally high temperatures may be lethal to both seeds and buds, whilst sub-lethal temperatures may impair or enhance regeneration.

In general a successful heathland management fire with temperatures below 400 ^{0}C will remove all above ground vegetation but leave the rootstocks unharmed, but only a fire with canopy temperatures above 500 ^{0}C will ensure the combustion of all vegetation (Webb 1986). But the stem bases of *Calluna* are killed if exposed to temps over 500 ^{0}C for more than one minute (Whittaker 1961).

Malik and Gimingham (1985) studied the effects of fire severity on 12 heathland species, other than *Calluna*, using 2-minute *ex situ* fire treatments of 400, 600 and 800 ^{0}C (replicated over five individual plants). Of the species studied only *Juniperus* (a UK BAP Priority Species) was completely sensitive to fire because of its unprotected above-ground buds. This species was killed off by a 800 ^{0}C fire and only feeble growth occurred after the 600 ^{0}C treatment. Observations suggested that regrowth may only occur if some of the basal green branches remain alive, but as the species has thin bark, it is unlikely that these would survive hotter fires. Hobbs *et al* (1984) suggest that this in part explains the scarcity of *Juniperus* in many British heaths compared with those in Europe. After most fires *Juniperus* regeneration will need to be from seed, and seedling establishment is unlikely where *Calluna* vegetative regrowth is rapid. Its lack of seed dispersal into recently burnt areas (Miles and Kinnaird 1979, cited in Hobbs *et al* 1984) also probably prevents it from establishing in areas where *Calluna* regrowth is by seed and therefore slow. But MacDonald (pers comm 2003) considers that this may be an oversimplification because there is not a clear negative correlation between burning management and the occurrence of *Juniperus* in the UK. Indeed, in the Scottish highlands, *Juniperus* is most abundant and widespread where burning management is common. It also appears that *Juniperus* can become established in recently burnt patches when there are nearby seed producing stands. Colonisation may also be aided by seed dispersal by birds. Overall it is likely that the occurrence of *Juniperus* is affected by its undoubted sensitivity to fire, but also its requirement for ground disturbance for regeneration and then its need for lack of disturbance during establishment.

In contrast Mallik and Gimingham found that all ericaceous species, notably *Erica tetralix*, sprouted vigorously after burning. But in all cases sprouting was least after higher temperatures; *Empetrum nigrum* being particularly adversely affected by the 800 ^{0}C treatment. Recovery and subsequent performance in *Erica cinerea* and *Vaccinium myrtillus* appeared to decline as fire temperatures increased. However, *Arctostaphylos uva-ursi*, *Empetrum nigrum* and *Erica tetralix* apparently responded most to the 600 ^{0}C treatment, whilst *Succisa pratensis* regrowth was greatest at 600 ^{0}C and 800 ^{0}C. Regrowth appeared to be unrelated to the range of fire temperatures tested in *Genista anglica*, *Pyrola media*, *Vaccinium vitis-idea* and the two grasses, *Deschampsia flexuosa* and *Festuca ovina*.

Whitaker and Gimingham (1962) found that seeds of *Calluna* were killed by its exposure to 200 ^{0}C but that below this level effects depended on the period of exposure. Short exposure to temperatures of 40-60 ^{0}C stimulated germination whilst longer exposures reduced germination. Mallik and Gimingham (1985) found that germination appeared to be stimulated by heat in several species (*Erica tetralix*, *Genista anglica*, *Hypericumm pulchrum*) but many showed complex interactions between temperature, length of exposure and pre- or post-treatments.

As described in Section 3.2, the temperature of a heathland fire is primarily dependent on the amount of woody *Calluna* fuel which is proportional to the average height of the stand (Hobbs and Gimingham 1984b). Maximum soil surface temperatures of over 600 ^{0}C were recorded for several mature and degenerate stands (although these varied greatly). Thus there is certainly some potential for hotter fires to inhibit, at least to some extent, the vegetative recovery of *Calluna* and some other sensitive species. But actual impacts on protected buds, stools, rhizomes and seeds are more difficult to ascertain, as it is not clear how laboratory test results relate to temperatures experienced by seeds in the soil. More research work needs to be done on this.

Nevertheless, it is clear that burning older stands has a double risk: the regenerative capacity of old stands is low (see above) and the hot fires that they produce can further reduce their capacity for vegetative regrowth and damage the seedbank too.

But an important factor that may mitigate against the impacts of hot fires in older *Calluna* stands is the presence of moist moss mats and deep litter layers, which help to limit soil temperature rises and protect seedbanks and buried stools from burning (see Section 3.2). The presence of wet humus and soils may be particularly important as soil temperatures in fires are limited to 100 ^{0}C where water is present, although it should be noted that surface layers may dry out and burn whilst lower layers remain wet (as is the case in smouldering peat fires).

Wildfires can lead to exceptionally high temperatures which can ignite dry peat and humus layers (see Section 4.1.1), leading to the lose of all vegetation and seed banks. In such cases revegetation may be very slow and will not follow the predicted pattern described, but will be most influenced by the colonisation ability of species and the substrate conditions. Typically revegetation is very slow and in such circumstances initially restricted to certain bryophytes. For example, recolonisation of severely burnt areas of Rosendale Moor (see Section 4.1.1 for background) was initially dominated by bryophytes, including *Ceratodon purpureus*, the main pioneer on ashed surfaces, and *Dicranella heteromalla* and *Pohlia nutans*, the dominants on intact peat surfaces (Maltby *et al* 1990). These were succeeded by *Polytrichum* species, with *P. piliferum* dominant on the drier substrates and *P. commune* dominant on wetter areas, often with residual peat. After 8 years, 65% of the area remained occupied by bryophyte communities, only 6% of the area supported vascular species and only 2% consisted of *Calluna* dominated communities. Nearly 30% of the area remained unvegetated.

However, wildfires do not always lead to such long-lasting damage and their outcome can be variable. For example, a large fire on western heath on Trendlebeare Down on Dartmoor, led to the complete loss of vegetation over a very large area. But soil damage did not appear to be severe and secondary erosion was avoided by a relatively rapid recovery of the *Ulex* spp. Maltby et al (1990) also report that after apparently catastrophic summer fires in 1976 on Hartland Moor in Dorset and Glasson Moss in Cumbria, vegetation similar to that present before the burn had re-established within a decade. Similarly, a study by Bullock and Webb (1995) observed that what appeared to be large and intense fires on lowland heaths did not always lead to the disastrous impacts feared immediately after the event.

Timing of burning

Autumn burns on dry heaths are usually less intense than spring burns, since the vegetation and litter have a higher moisture content in the autumn (Hobbs & Gimingham 1987). Similarly, the intensity of burning is lower when proceeded by wet weather. Thus the timing of burning in relation to season and pre-fire weather conditions can often be used as a means of controlling fire temperatures (D. Coates pers comm 2002). For example burns that are set in cold weather and at the start of a dry spell may create 'cold' fires that remove standing vegetation but leave mosses etc undamaged.

An experimental comparison of autumn and spring burns in north-east Scotland found that *Calluna* regenerated more successfully after autumn fires (Miller and Miles 1970). This was probably, at least in part, because potentially competitive species such as *Eriophorum vaginatum* and *Scirpus cespitosa* are less competitive at this time of year (Mowforth and Sydes 1989).

One of the arguments against burning in September (ie before the start of the legal burning season on 1st October) is concern over the regeneration response from heather burnt before it is properly dormant (Phillips 1991b). But from anecdotal evidence from accidental fires, Phillips considers that burning in autumn does less damage to *Calluna* than spring fires when growth has begun.

But a potentially significant detrimental impact of autumn burning is increased erosion, because bare or sparsely covered soil will remain over winter. This bare soil will be particularly susceptible to erosion due to the presence of needle ice, freeze-thaw cycles and typically high rainfall. There may also be disadvantages from burning in autumn if regeneration is likely to be by seedling establishment. This is because there is an increased risk that any seedlings germinated in autumn will be uprooted by needle ice over the winter.

Interactions with grazing

Interaction between burning and grazing will tend to depend on the vegetation types affected, the fire return frequency, size and locations of burns, and the number and type of stock grazing the burnt areas and the seasonal timing of grazing. However, in general grazing and burning on heathland affects the vegetation in similar ways. For example, both increased burning frequency and grazing intensity tend to change vegetation from *Calluna* dominated communities to grasslands Thus grazing tends to accentuate the impacts of burning.

Few studies have examined specific interactions between grazing and burning. But early work found that there was a strong effect of grazing pressure on the balance between *Calluna* and *Erica cinerea* in post-fire heathland communities (Gimingham 1949). *Erica* seedlings were much more abundant in the absence of grazing, and as grazing pressure increased the relative proportion of *Calluna* in the post fire community increased. It has since been observed that sheep grazing can have a substantial impact on post-fire heathland development by influencing the rate of *Calluna* recovery (Grant 1968, Grant and Hunter 1968, Grant 1971). In particular heavy grazing keeps *Calluna* short and may allow grasses such as *Deschampsia flexuosa* to persist for long periods.

It has been suggested that the losses of heather moorland to grassland which have occurred in many areas has been due to large increases in the numbers of sheep and deer in Britain in recent years, often in combination with poor burning practices (Sydes and Miller 1995). The resulting prolonged and intensive grazing pressures have caused a switch in habitat because dwarf shrubs are less able to withstand repeated heavy grazing than grasses.

But paradoxically, when grazing is heavy regular management by burning may be particularly important to maintain heather dominance. Large areas of tall, old heather are generally little grazed, and therefore grazing animals tend to concentrate on areas of younger heather that are readily accessible and more nutritious. Thus a decline in burning may reduce the sustainable stocking density and thereby trigger or exacerbate overgrazing problems. Furthermore, if burning rotations are lengthened such that *Calluna* stands are burnt at, or beyond, the late mature phase then regeneration will be largely by seed. With sparse regeneration seedlings are readily accessible to grazing animals and this may prevent successful recolonization of heather under such conditions. Burning in patches is also important as this can help encourage sheep to disperse and graze more widely across their range (Shaw *et al* 1996).

But as previously noted more research is required into the interactions between grazing and burning.

Scrub control

As discussed above, scrub may develop on suitable ground in areas with potential tree colonists where grazing levels and fire frequencies are low. Scrub may therefore be burnt as part of heather management or with the specific intention of its removal.

On the south-west moors (Bodmin Moor, Exmoor, the Quantocks and Dartmoor) gorse *Ulex* spp scrub / heath habitats, such as H4 *Ulex gallii – Agrostis curtisii* heath and H8 *Calluna vulgaris-Ulex gallii* heath, are common on the upland fringes. Furthermore, it has been suggested that milder winters in the region have encouraged the spread of gorse leading to large stands of tall scrub, which are impenetrable to livestock and hinder recreation uses. These habitats are therefore often burnt as an attempted control measure. However, although the vegetation is effectively temporarily removed, burning is usually ineffective in reducing *Ulex* in the post-fire community. Like most broad-leaved woody species, *Ulex*, will unless very old, sprout vigorously from the base (Miles 1987). Temperatures in *Ulex* fires do not appear to have been measured, but it is likely that they are very hot as mature stands will contain large amounts of woody fuel. Such hot fires may reduce vegetative regeneration capabilities in *Calluna* and other heat sensitive species (see above), damage seedbanks and cause high nutrient losses. But on the other hand, although such fires can produce a lot of heat most is likely to be released in the canopy which is significantly higher above the ground than in *Calluna* and other dwarf shrub fires. Thus the ground fire intensity may not necessarily be as high as one might think, particularly as one can sometimes find unburnt or little burnt material on the ground under burnt *Ulex* stands.

Many other upland woody species such as *Betula* spp, *Sorbus aucparia*, Salix spp, *Crataegus monogyna*, *Quercus petraea* and *Q. robur* may also be burnt during heather, bracken or grass fires. These may be cut-back by fires, but generally resprout (Miles 1987). It has been suggested that *Betula nana*, a rare species in England with outlying populations in the North Pennines and border uplands (Drewitt and Manley 1997) is intolerant of repeated burning and grazing (Aston 1984, De Groot *et al* 1997). However, MacDonald (pers. com. 2002) states that “field observations show that it tends to resprout readily after burning. But is heavily browsed and burning would tend to make it more conspicuous and attractive to browsers.”

The presence of the species in wetter areas may reflect reduced burning impacts, but may also be due to lower browsing pressures, as bogs tend to have lower densities of sheep and deer.

Juniperus (see above) and *Pinus sylvestris* are probably the only native fire sensitive woody species present in the British uplands. However, as noted above, the occurrence of *Juniperus* is also likely to be related to colonisation abilities, and the recent *Juniperus* decline in England appears to be related to heavy grazing pressure and consequent low recruitment and establishment of seedlings (Drewitt and Manley 1997).

5.2.2 Wet heath

Wet heaths are most commonly found in the wetter northern and western parts of Britain and, in favourable condition, are dominated by mixtures of *Erica tetralix*, *Trichophorum cespitosum*, and *Molinia*, over an understorey of mosses often including carpets of *Sphagnum* species. The impacts of fire and grazing on wet heaths has been reviewed by Shaw et al (1996). In general it appears that little specific research has been carried out on the impacts of burning on this habitat compared to dry heaths (see above); although a PhD study was undertaken of burning impacts on wet heaths on the Isle of Sky, in western Scotland (Currall 1981), but the results of this remain unpublished. However, many of the key factors affecting impacts of burning on dry heaths, such as pre-fire species composition, stand age, fire intensity and grazing impacts appear to apply to this habitat as well to some extent.

But their response to fires is less dependent on the condition of *Calluna* in the pre-fire community. In fact because *Erica tetralix* tends to regenerate more quickly than *Calluna* in wet heaths (and bogs) it can sometimes become unusually abundant (especially where there is also quite high grazing pressures as *Calluna* will be preferentially browsed). Currall (1981) concluded that post fire successions in wet heaths in north west Scotland typically follows three phases. Firstly, there is an early graminoid phase, which is dominated by species that are able to rapidly recover or colonise bare ground after fires. The actual species present depends on the pre-fire community composition, but the phase is characterised by a relatively high number of, mainly graminoid, species and a high proportion of bare ground. Typical dominant species of this phase include species such as *Carex echinata*, *Eriophorum angustifolium* and *Juncus squarosus* in wetter areas, and *Carex panacea*, *Nardus stricta* and *Agrotis* spp and *Festuca* spp in drier areas. Herbs such as *Potentilla erecta* may also be present.

The second phase, is the dense graminoid phase, and results from the establishment of dense growths of *Molinia* or *Trichophorum cespitosum* and a reduction in bare ground, which restricts the opportunity for new species to arrive. The phase may last 8 – 12 years, over which litter builds up and increasing shading gradually eliminates the shade sensitive species, so that the number of species gradually falls. Because *Erica tetralix* is able to recover quickly from fires it may reach its peak cover in this phase, but other ericaceous species remain less common.

Calluna and most other dwarf shrubs become dominant in the third (ericoid) phase, typically about 15 years after the fire. At the same time the cover of most graminoids decline, whilst bryophytes develop under the canopy. With further time, *Potetentilla erecta* and *Eriophorum vaginatum* may reappear as gaps form in the heather canopy.

As with dry heaths, the impacts of long-term serial burning on wet heaths appears to vary according to the severity and frequency of burning. Shaw et al (1996) suggest that drainage and burning of M15 *Trichophorum cespitosum – Erica tetralix* wet heath tends to shift vegetation away from *Sphagnum* rich sub-communities to *Calluna*-rich and heathy *Cladonia* sub-communities. But occasional grazing and burning may be important in the maintenance of M16 *Erica tetralix – Sphagnum compactum* wet heaths where they occur on upland fringes, and according to Shaw *et al* (1996), burning has a marked effect on the floristic variation and local patterns of dominance and richness of the associated flora. Controlled burning to rejuvenate pioneer and building phase *Calluna*, may be responsible for the predominance of *Calluna* in the M16d *Juncus-Dicranum* sub-community type in north-east Britain.

But severe burning and grazing can convert M15 into various forms of acid grassland (eg U4 *Festuca ovina–Agrostis capillaris –Galium saxatile*, or U5 *Nardus stricta–Gallium saxatile*). Many areas of wet heath are also frequently burnt, eg as noted on Skye (Currall 1981) and Dartmoor (Goodfellow 1998), to remove the rapid build up of *Molinia* litter. As in dry heaths, such frequent burning dehabilitates *Calluna* and favours more rapidly recovering species. Thus frequent burning has been observed to lead to an impoverished form of M16 wet heath (Drewitt and Manley 1997) and an increase in *Molinia* and *Trichophorum cespitosum* (Currall 1981, Miles 1987, 1988). Currall also notes that burning conditions on Skye are often too wet for effective burns, and when burns are carried out in such circumstances they burn back *Calluna* and other dwarf shrubs but are ineffective in removing the wet grass litter below. This further encourages the spread of *Molinia* at the expense of *Calluna.* Gardner et al (2001) note, from a study of the effects of spatial variation in sheep grazing on *Calluna - Molinia* interactions, that great care is needed with heather burning on wet heath to ensure that *Molinia* does not gain dominance. In fact many areas of M15 are also likely to have formed from frequently or severely burnt and grazed areas of blanket mire (see Section 5.7).

Wet heaths are often rich in bryophytes and lichens, and although they may be protected by wet conditions, Shaw et al suggest that some, such as *Cladonia* species, may be lost through frequent fires as they are normally slow to recover (see also Figure 5.4).

5.3 Calcareous grassland

As noted in Table 1.1 two Annex 1 Habitats fall under this Broad Habitat category (and associated *Juniperus* scrub) and Upland Calcareous Grasslands has been identified as a UK BAP Priority Habitat.

Very little information has been located in this review on the extent or practice of burning calcareous grasslands, and very few scientific studies have investigated its impacts. However, one study, although now old, describes the effects of fire on calcareous grasslands in the Derbyshire Dales (Lloyd 1968). At the time of the study, many of the slopes of the dales were under-grazed and had formed rough grasslands that were prone to accidental fires. These generally occurred in spring when the winter accumulation of plant material had dried out. Three grassland types were studied: *Festuca ovina-Helictotrichon pratense* communities on well drained rendzina soils, *Agrostis tenius-Anthoxanthum odoratum* communities on well drained soils (though with a high proportion of non-calcareous loessic deposits) and *Agrotis tenius-Festuca rubra-Holcus lanatus* communities on moist heavy clay soils.

Lloyd found that fires resulted in an increase in the frequency of short-lived species with the ability to regenerate from ground-level or below. For example, in the *Festuca –Helictotrichon* community, fire favoured the densely tussocked *Helictotrichon pratense* in comparison to the more loosely tussocked *Festuca ovina*. The open turf conditions created by the fire also led to a temporary increase in the frequency of annual and biennial species, which are less common in the grazed and unburnt grasslands. Invading woody plants were severely checked, but rarely killed outright.

A later study by Lloyd (1971) examined the effects of fire on nutrients in the soil and vegetation (previously described above in Section 4.2). This found that there was no evidence that observed high losses of phosphorus and potassium were limiting vegetation growth in the first two years after burning. In fact there were slightly higher levels of phosphorus in the living tissues of plants in burnt areas compared to unburnt areas, and recovery of the vegetation in terms of standing crop and chemical composition was complete after three to four years. Overall, the apparent substantial nutrient losses from the fire therefore appeared to have no overall effect on the vegetation.

5.4 Acid grassland

Acid grasslands are some of the most extensive upland habitats in England and comprise a range of typically species-poor NVC communities U1-U6 (Rodwell 1992, Drewitt and Manley 1997, Backshall *et al* 2001). As described in Section 2.2.1, many acid grasslands have a long history of burning management for agricultural purposes, the principal purpose usually being to remove litter and stimulate fresh nutritious spring growth. Burning of these habitats is often carried out by large and uncontrolled fires and at frequent intervals, eg on the west coast of Scotland (McVean and Ratcliffe 1962, Currall 1981) and has been a common practice for centuries on Dartmoor (Goodfellow 1998). In some areas grasslands are burnt annually, such as on the *Molinia* or *Agrotis-Festuca* grasslands of Exmoor(Miller *et al* 1984).

The effects of fires on acid grasslands appear to have been relatively little studied despite their long history of frequent and large-scale burning (and the likelihood that many upland grasslands have developed as a result of the burning of heaths and bogs). The studies that have been carried out suggest that in the absence of burning litter builds up, which can smother other vegetation and lead to increasing dominance of species such as *Molinia*. Studies of lowland heathlands in continental Europe also support the view that a build up of litter, and associated organic matter accumulation and nutrient mineralization, encourages the replacement of ericaceous shrubs, such as *Calluna* and *Erica tetralix*, with grasses such as *Molinia* and *Deschampsia flexuosa* (Aerts 1990, Berendse *et al* 1994). Thus removal of litter and possible nutrient depletion (see Section 4.2) associated with burning may help to maintain dwarf shrub communities. But too frequent burning perpetuates fire resistant species such as *Molinia* and (as described in Section 5.1) dehabilitates *Calluna* because it is burnt-off before it reaches maturity (McVean and Ratcliffe 1962, Grant *et al* 1963, Miles 1971, Currall 1981, Miles 1987). *Molinia* is able to rapidly recover and out-compete other species after burning because its tiller buds are situated deep within the basal sheaths of its tussocks and are unaffected by most fires (Grant *et al* 1963).

But the quality of forage is reduced where *Molinia* becomes dominant because it produces large quantities of tussocky litter which interferes with the ability of livestock to graze fresh growth. Hence further burning is required to remove the litter and provide an accessible fresh spring 'bite'. Thus a downward spiral is initiated where an increase in *Molinia* leads to more frequent burning which in turn further increases *Molinia* dominance to the long-term detriment of the agricultural value of the grassland (Currall 1981). The problem is exacerbated by the large areas that are typically burnt each year. These areas often produce more fresh spring growth than can be grazed by the livestock present (as their numbers are limited by forage availability at other times of year) causing over-production of *Molinia* and further litter build up (Grant *et al* 1963). Thus as observed by Ball (Ball 1974) on the Isle of Rhum, *Molinia* tends to spread if burning is followed by inadequate summer grazing.

With such frequent burning, there is the added likelihood of long-term nutrient depletion (and consequent declines in forage quality), but no studies appear to have attempted to investigate this.

It is has been suggested that long-term frequent burning has been one of the principal causes of an expansion of *Molinia* grasslands on the south-western moors. For example, much of the deep peatlands of Dartmoor are covered by large expanses of *Molinia* dominated grassland (Ward *et al* 1972). These are presumed to be former blanket bogs (and indeed bogs and intermediate grassland-bog habitat types remain) and their conversion to grassland is thought to have taken place over several centuries of traditional burning. Citing Vancouver (1808) , Ward *et al* suggest that *Molinia* was a conspicuous species at the time of Vancouver's account and that ericaceous species such as *Calluna* and *Erica tetralix* were not.

But a recent palaeoecological study of Exmoor suggests that *Molinia* dominance may not be simply the result of burning or other management practices (Chambers *et al* 1999). The study indicates a greater antiquity and former abundance of *Molinia* than is often appreciated and suggests that, over the past millennium, vegetation dominance has alternated between *Calluna* dominated vegetation and grass moor, both containing at least some *Molinia*. Furthermore there is some circumstantial evidence that changes in these vegetation assemblages have coincided with climate shifts and may, therefore, be determined by them.

Recent increases in *Molinia caerulea,* as well as *Nardus stricta*, on many uplands in England have also been attributed to other inappropriate management regimes including overgrazing and increased moorland drainage as well as changes in burning practices (ADAS Consulting Ltd 2001).

The impacts of the conversion of bog and heath habitats to acid grasslands as a result of frequent burning have not been quantified. But the areas affected are likely to be substantial and the effects on biodiversity will be highly detrimental as bog and heath habitats are generally of higher conservation value. Also, because grassland burning is typically carried out over large areas in one go, then burning tends to reduce large-scale species and structural diversity. Grassland burning does not produce the mosaic of differing vegetation as on carefully burnt grouse moors, but instead creates uniform expanses of even aged and species-poor habitat.

The impacts of frequent burning are also all the more serious because increases in *Molinia* dominance are difficult to reverse without direct intervention. A considerable amount of research effort has focussed on moorland management and possible methods for *Molinia* control (Anderson *et al* 1997, Ross *et al* 2000, Todd *et al* 2000, Anderson 2001) see Anderson (2002) for brief review. These studies and treatment trials have revealed that it is possible to restore *Molinia* dominated upland areas to heathland using various combinations of burning, herbicide treatment, cutting and grazing. But the methods are expensive because of the large areas that need to be treated. Costs for spraying burning and seeding large areas have been £225 ha^{-1}, and it may be necessary to apply follow up treatments as well (Anderson 2002). There are also likely to be large areas of the uplands where it will be impractical and undesirable to carryout such interventions.

5.5 Bracken

Bracken *Pteridium aquilinum* occurs as a scattered component of many heathland and grassland habitats, but it also commonly occurs in virtually pure stands where it forms a distinctive habitat in its own right. Such bracken stands have spread considerably in the UK, with some estimates indicating that they have doubled their extent in the last century

(Backshall *et al* 2001). It has been suggested that its spread is a result of its unpalatability to livestock (exacerbated by a recent reduction in cattle stocking rates on hills in favour of sheep), inappropriate burning practices and climate change, as the recently wetter and warmer winters favours the species (Marrs and Pakeman 1995, Drewitt and Manley 1997).

But, the extent to which the spread of *Pteridium* is due to burning is unclear as there is evidence that the likelihood of post-fire *Pteridium* invasion very much depends on the vigour of the regeneration of the original vegetation. Where observers have returned to burnt areas they have found that where *Calluna* regeneration has been vigorous it has sometimes been able to compete well with the *Pteridium*. Published data demonstrating this are few, but MacDonald (pers. com. 2003) reports that Sykes (1988) studied the vegetation changes on the site of a ground fire (with small amounts of crown fire) which swept through a pinewood on the steep slopes of Ben Shieldaig in Torridon. Twelve years after the fire Sykes reports that *Molinia* and *Pteridium*, both of which had reached a peak two years after the fire, were continuing to decline, whilst *Calluna* and bryophytes, which had almost been eliminated by the fire, were continuing to increase. Furthermore he noted that the dominance of *Pteridium* had been reduced markedly as a result of vigorous growth of *Calluna*.

A more recent study in the Quantock Hills, in south-west England, also indicates that averaged over time the effects of burning management of heath is more likely to slow the spread of *Pteridium* than to encourage it (Ninnes 1995). And there is also some evidence that *Calluna* can out-compete bracken in specific circumstances from experimental work at the Department of Plant and Soil Science, in Aberdeen. This has shown that *Calluna* is the superior competitor for water under drought conditions (Gordon *et al* 1999).

Pteridium is most likely to invade heathlands where the regeneration of *Calluna* is slow and *Pteridium* rhizomes are present in the soil or adjacent areas (Watt 1955). According to the studies reviewed above, this is most likely to occur when burning removes either the pioneer or degenerate stage of *Calluna* vegetation, or after severe fires which damage the ability for vegetative regrowth. Thus, too frequent fires (eg every couple of years), or hot fires and fires in *Calluna* stands at the late mature or degenerative stage are likely to encourage *Pteridium* invasion.

Once established, *Pteridium* is highly competitive and very difficult and expensive to control. It cannot be controlled by fire alone, but requires other treatments such as herbicide applications (see Backshall *et al* 2001 for details).

5.6 Fen, marsh and swamp

As described in Section 1.3.7 fen, marsh and swamp habitats are common in the uplands and include a variety of vegetation communities including *Molinia* meadows (NVC type M26), transition mires, quaking bogs, alkaline fens, flush mires, flushes and springs. They include 5 Annex 1 listed habitats (Table 1.1).

Fen, marsh and swamp habitats that are normally water-logged throughout the year are unlikely to be directly affected by fire, unless this coincides with drought conditions. Such wet habitats tend to be fire refugia. For example, wet flushes were little effected by the severe wildfire on Rosendale Moor on the North York Moors (Maltby 1980, Maltby *et al* 1990).

Some of these habitats may, however, be seasonally dry, and during such periods may be burnt where they occur in patches within drier moorland habitats that are subject to burning management. But they would normally only be expected to be sufficiently dry to burn outside the legal burning season. However, in some years they may be sufficiently dry to sustain burning at the beginning or end of the burning season. On such occasions burning is likely to be accidental, as there is little to be gained from burning such habitats. But as wet ground is often used as a natural firebreak in controlled burns, such accidental burning may be more common than initially supposed in dry years. Even such occasional burns on these habitats could be detrimental and have long-term impacts, particularly on some bryophytes and other fire sensitive species. But in general the most likely cause of direct significant damage or complete loss would be wildfires during dry weather, rather than prescribed burning under appropriate conditions.

Fen, marsh and swamp habitats may be indirectly affected by fires, the most likely serious impact being as a result of large scale erosion. This could cause complete loss of the habitat if landscape scale impacts occur, such as after large scale fires. Smaller scale erosion initiated by prescribed fires may also cause impacts through increased sediment deposition, which may swamp existing flora and possibly change nutrient regimes.

No scientific studies have been found in this review that assess the potential impacts of burning on upland fens, marshes and swamps. Given the high conservation importance and scarcity of several of these habitats research into such impacts is required.

5.7 Bogs

Bogs (or ombrotrophic mires) are abundant in upland areas and support a variety of normally peat-forming acidophilous species, in particular bog-mosses *Sphagnum* spp, *Eriophorum* spp and *Erica tetralix*. They are of considerable conservation importance in the EU and the UK and include 2 Annex 1 Priority Habitats (Table 1.1), whilst blanket bog is also a UK BAP Priority Habitat.

Several of the factors determining the impacts of burning bog habitats are common to those affecting heaths, such as pre-fire vegetation composition, stand age, fire intensity and post-fire grazing pressure (see Section 5.2). But there are two key ecological differences between bogs and heathlands that affect their response to these factors. Firstly, the typical *Calluna* growth cycles seen on heathlands do not normally occur on bogs (unless they are degraded), because layering occurs as the stems are buried by growing *Sphagnum* (Rawes and Hobbs 1979, MacDonald *et al* 1995). As a result the age of the above ground stems are uneven and comparatively low, and degeneracy does not occur. Instead the bog is said to be in a 'steady state' (Forrest 1971). Secondly, the growth and productivity rates of *Calluna* on bogs is much lower than on heathlands.

Studies of burning at Moor House NNR in the north Pennines found that as a result of their fire resistance, *Eriophorum* spp regenerated more rapidly than *Calluna* and became dominant for up to 15 years (Rawes and Hobbs 1979, Hobbs 1984). After which *Calluna* regained its dominance in combination with *Eriophorum* and *Sphagnum* species. Thus short rotation burning favours *Eriophorum vaginatum* because *Calluna* is relatively scarce for the first 10 or so years and can only produce limited vegetative regrowth after a subsequent fire, whilst the establishment of *Calluna* seedlings is prevented by the *Eriophorum vaginatum*. Conversely burning at longer rotations (eg every 20 years) favours *Calluna* as it is by then dominant in relation to *Eriophorum vaginatum*. Similar results were found by Currall (1981) in wet heaths (Section 5.2.2).

Where *Eriophorum* spp are sparse some increase may be beneficial since the leaves (in winter), and the developing flower heads (in early spring), can be important sources of food for moorland livestock, game and some rarer species of wildlife (eg black grouse and large heath butterfly *Coenonympha tullia*).

Burning is also known to favour *Rubus chamaemorus* in bogs as this species can spread rapidly from underground rhizomes, especially in the absence of competition. This species was also seen to benefit from burning at Moorhouse (Hobbs 1984). In other areas where *Molinia* replaces *Eriophorum* spp frequent burning encourages the dominance of this species (see Sections 5.2.2 and 5.4 above).

Anderson (1997) notes in the Peak District, that after well controlled management burns *Calluna* dominated vegetation on thinner peaty podsols tends to recover relatively quickly. But on thicker peats, bog vegetation is dominated by *Eriophorum* spp and dwarf shrubs such as *Empetrum nigrum* and *Vaccinium myrtillus* together with *Calluna* and *Rubus chamaemorus* in places. When burnt, even superficially, this type of vegetation does not always recover quickly, with *Empetrum* being the most likely species to fail to recover.

Hobbs (1984) questioned the use of management burning on bogs where *Calluna* develops a steady state in which it is constantly rejuvenated. He also pointed out that management benefits for red grouse are unlikely to occur because the initial loss of *Calluna* from fires would probably outweigh the improvement in *Calluna* productivity ; although there were no data to support this at the time. Coulson (1992) also noted that the loss of *Calluna* habitat over the several years that it takes to re-establish after burning may be significant (eg as much as 25% of area), yet is generally disregarded by grouse moor managers. Subsequent advice for moorland managers from the Game Conservancy Trust suggests that the burning of blanket bogs should be a low management priority for red grouse (Hudson and Newborn 1995).

Studies have also shown that there is a general decline in plant species-richness on burnt areas of blanket bog (Hobbs 1984). *Sphagnum* mosses in particular have often been regarded as being especially susceptible to burning (eg Ratcliffe 1964, Rowell 1990). Whilst Rodwell (1991b) cites Pearsall (1941) in suggesting that *Sphagnum* is less effected from the fire itself, than from the subsequent effects of exposure to drying, and the indirect effects of burning and drying on the peat.

However, MacDonald (2000), citing several sources, states that field observations indicate that *Sphagnum* mosses are actually quire good at surviving normal, fairly low intensity management fires. This may be because fires tend to pass through the vegetation layers above the *Sphagnum* (Daniels 1991, Barkman 1992). Bryophytes and lichens may appear dead after the passage of a fire, but in fact many have some capacity to survive and regenerate if the fire is light with minimal smouldering and no combustion of the mor humus layer (Schimmel and Granström 1996).

An experimental study in northwest Scotland of the recovery of *Sphagnum* after fires found that, although effects were highly variable, the perception that fires are always extremely detrimental to the *Sphagnum* layer of a bog was unsupported (Hamilton 2000). A sizeable proportion of the *Sphagnum* recovered in the first two years after the fires. However, the damage to *Sphagnum* increased with increasing amounts of overlying *Calluna*, presumably because this created hotter and more intense fires

Shaw et al (1996) state that hummock forming species may be susceptible to fire, particularly when dry and cite Slater (1976) in suggesting that fire may have caused the decline of the rare *Sphagnum imbricatum* in Cors Fochno, Wales. But a later study suggests that this may not be the case, as fire was probably not the cause of a decline in the same species at Coom Rigg Moss NNR (Chapman 1991). In wet conditions hollows would normally be water filled and in such circumstances would be fire refuges. But MacDonald (pers. com. 2003) suggests that in dry conditions tight hummock forming species may be able to survive fires and the subsequent exposed post-fire conditions better than species associated with hollows, because they retain more moisture. MacDonald states that he has observed that in dry summers "peat mosses of the hummocks, *Sphagnum papillosum* and *S. magellanicum*, were still fresh and moist whereas *S. apiculatum*, and in particular *S. cuspidatum* in the hollows, were bleached and dry as paper". Indeed (Müller 1965 cited in, Barkman 1992) notes that in a living bog, fire affects the hollows more than the hummocks, and that high hummocks may burn out completely but low hummocks of *Sphagna cymbifolia* are not (or hardly) affected.

Nevertheless, there is certainly a risk to *Sphagnum* and other bryophyte species from burning bogs, especially in dry conditions and if tall heather is present that may create hot fires. But, with current scientific information it appears impossible to reliably identify species of lichen, bryophyte and fern that are actually significantly impacted by burning. Although species that occur in drier habitats may be more frequently exposed to burning, those of wetter bogs may be less fire resistant.

Wildfires, large prescribed fires and, to a lesser extent, back-burning are likely to be especially harmful to bogs as they are likely to trigger erosion. There is also a risk of bog fires igniting peat layers in dry conditions. This may cause removal of the surface layer of peat which results in poor vegetation regeneration, which in turn makes such areas highly susceptible to severe erosion (see Section 4.3). The supplement to the *Muirburn Code* notes that "conditions that permit good control of fires are exacting and infrequent on peat ground: either much material is left unburnt, and heather regeneration is poor, or the effects are too intense and the underlying peat is exposed" (Anonymous 2001c).

Losses of nutrients from burning on blanket bog may also be more detrimental than on heathland because the vegetation is isolated from nutrient sources in mineral soils by an intervening layer of peat (Coulson *et al* 1992). On the other hand, as noted previously, there is insufficient information available to indicate whether this is important or not as low nutrient conditions are a characteristic of blanket bogs (as well as heaths).

Thus in conclusion, although evidence is patchy, it appears that the burning of bogs is generally unnecessary and often detrimental to its characteristic vegetation, and possibly some species of particular importance, as well as peat soils. As noted in Section 4.3, there is also a high risk of erosion following fires on bogs (particularly following peat ignition) which may lead to further vegetation impacts and significant losses of sequestrated carbon. Impacts from prescribed burning are most likely to be high when managers aim for 'clean burns' as these will be carried out in dry conditions when there is a high risk of damaging or burning the underlying peat. As a result of these burning impacts, it is generally recommended that bog habitats are either not burnt at all or on very long rotations (eg Rowell 1988, Coulson *et al* 1992, MAFF 1994, Thompson *et al* 1995b, Anonymous 2001c). Further information on the impacts of burning bog habitats is provided in Shaw *et al* (1996).

5.8 Montane habitats

These habitats occur exclusively in the montane zone and include a range of vegetation types such as prostrate dwarf shrub heath, snow-bed communities, sedge and rush heaths, and moss heaths (see Section 1.3.9). Two montane habitats are listed on Annex 1 of the Habitats Directive: alpine and boreal heaths, and siliceous alpine and boreal grassland (Table 1.1).

Given their altitude most montane areas are unlikely to be affected by burning. However, it is possible that some areas of montane habitat, in particular the dwarf shrub communities, may be affected by occasional accidental fires in dry summer conditions. Although no data appear to be available on the extent of such fires, it is probable that they are rare, if they occur at all.

No scientific studies appear to have been carried out on the impacts of fire on montane habitats in Britain. One may speculate that if fire does affect these habitats, impacts may be determined by similar factors to those that influence analogous lower altitude habitats (eg H12 *Calluna-Vaccinium* heath). But it is at least likely that impacts would be exacerbated by the slower growth rates and poorer, shallower soils that typify many montane habitats. If fires are infrequent in such habitats then *Calluna* stands are likely to be old with low capacities for vegetative regeneration after burning. In which case *Calluna* regeneration will rely on seedling germination, and seed production by declines with altitude, falling sharply above 600 m (Cummins and Miller 2002). However, this low seed production appears to be compensated for by a longer viability of soil sandbanks at higher altitudes.

Given the extreme weather conditions that affect montane habitats, fires are also more likely to trigger severe secondary erosion problems.

5.9 Inland rock

Rocky habitats are common and varied in the uplands. Many are of conservation importance at a UK and international level (Table 1.1). Of these the most widely distributed in the uplands of England is limestone pavement, which is a Habitats Directive Priority Habitat and also a UK BAP Priority Habitat.

The vegetation of rocky habitats is often rich in vascular plants, ferns, bryophytes and lichens, and includes many rare species. The extent to which burning affects these habitats has not been documented, but it is likely that in most rocky habitats fire is rare or non-existent, because in most there will be insufficient vegetative fuel to sustain a fire. Furthermore, many of the most valuable rocky habitats are damp (especially on north facing slopes) and any vegetative fuel that is present is unlikely to dry sufficiently to burn. Indeed, many of the characteristic bryophytes and lichens of these habitats are likely to be fire sensitive and slow growing, and therefore their presence is an indicator that these are fire refuges. For example, the U17 *Luzula sylvatica-Geum rivale* tall herb community grows on ledges and crags which provides protection from grazing and burning (Drewitt and Manley 1997).

Some rocky habitats may, however, be affected by burning where they consist of scattered rocks or small patches of rocky habitat amongst dry heathlands or similar habitats. But no information appears to be available on the extent to which this occurs, or its impacts on such habitats and their associated species.

5.10 Summary of impacts on key plant species

A summary of the known impacts on key plant species and species groups as described in the preceding sections of this chapter is provided in Table 5.2. These impacts reflect individual species abilities to survive and / or regenerate from fires and the indirect outcomes of effects on soil, hydrological and micro-climatic conditions, nutrient availability, competitors and other species in their associated communities. Impacts may also be influenced by grazing pressures, drainage and atmospheric pollution, but for the purpose of this summary, it is assumed that these factors do not have significant additive impacts, although as described above, in some case they may do so.

Table 5.2 Summary of impacts of burning management on key upland plant species and species groups

Species	Perennating organ & fire survival mechanism	Impacts
Betula nana	Stem bases, protected by litter*	Probably able to survive fairly low intensity management fires.
Juniperus communis	No survival mechanism*	Cannot survive fires or easily establish from seed in heathlands and is therefore normally absent where regular burning occurs and *Calluna* regrowth is vigorous. However, burning can expose bare ground for seedling establishment.
Calluna vulgaris	Stem bases, protected by litter*	Regenerates relatively rapidly after typical management fires, if burnt before the late mature phase. Re-establishes by seed from abundant long-lived seedbank if old stands are burnt or if hot fires damage basal stems. But seedling establishment is slow and may allow invasion by rhizomatous species. May not re-establish if burning is too frequent. Generally increases in abundance with long burning rotations (eg > 15 years) on bogs.
Empetrum nigrum	Buried branches*	May be susceptible to fires but if rhizomes are not destroyed then may gain temporary dominance in heathlands until overtopped by *Calluna*.
Erica cinerea	Stem bases, protected by litter*	Similar to *Calluna*, but possibly more susceptible to hot fires and favoured by shorter burning rotations of 6-10 years.
Erica tetralix	Stem bases, protected by litter*	Similar to *Calluna*, but favoured by shorter burning rotations of 6-10 years. May also be able to regenerate better in wetter habitats because its semi-prostrate lower branches are protected by *Sphagnum* and litter layers.
Vaccinium myrtillus	Underground rhizomes*	Rapidly regenerates after fires, and can attain temporary dominance. May out compete *Calluna* and remain dominant after hot fires or if old *Calluna* stands are burnt.
Vaccinium vitis-idaea	Underground rhizomes*	Less exposed to burning than *V. myrtillus* but response probably similar.
Eriophorum angustifolium	Tiller apices within leaf sheaves	Often benefits from periodic fires, as can rapidly recolonise burnt areas from rhizomes, but is later out-competed. May not survive post-fire conditions if significant changes in moisture and pH.
Eriophorum vaginatum	Tiller apices within leaf sheaves	Rapidly regenerates after fire and probably resistant to hot fires due to tussocky growth form. Temporarily dominates after fires in blanket bogs and can remain dominant if burning rotations are less than 10 years.
Molinia caerulea	Tiller apices within leaf sheaves	Can regenerate rapidly after fire and often dominates (sometimes with *E. vaginatum*) under frequent burning regimes.

Lichens	-	Most can survive fires to some extent, particularly when in large colonies. Crust forming lichens and some *Cladonia* species regenerate relatively rapidly after a fire (though less rapidly than mosses) but are out competed by *Calluna* and therefore these benefit from rotational burning. *Cladonia arbuscula*, *C. portentosa* and *Hypogymnia physodes* grow slowly under *Calluna* canopy and therefore are not favoured by fire, but can persist in burnt areas with long rotations.
Pioneer mosses	-	Establish quickly after a fire from vegetative fragments or spores but cannot tolerate competition with *Calluna* and are therefore favoured by short rotation burning.
Pleurocarpus mosses	-	Re-establish slowly (except for *Hypnum jutlandicum* if formerly abundant) from vegetative fragments and become most abundant under the *Calluna* canopy. Therefore, generally require long burning rotations or absence of fire, but can maintain presence in frequently burnt stands if fires are sufficiently light for fragments to remain alive and rapidly recolonise.
Sphagnum mosses	-	Often thought to be fire sensitive, but little evidence for this. Wet conditions may protect species from fires and some can regenerate from deep buried stems. Most impacts probably from peat damage and exposure to drying after removal of vegetation cover.

Note: * From Hobbs *et al* (1984)

6. Impacts on fauna

6.1 Invertebrates

6.1.1 Potential generic impacts

Relatively little research has been done on invertebrates in upland habitats and studies that have been conducted have focussed on relatively few areas, particularly with respect to blanket bogs (Coulson *et al* 1992). Northern heaths have been more widely studied, but in less detail. Nevertheless, these studies suggest that there are several key factors that influence upland invertebrate communities, which in turn may be affected by burning.

Firstly, the availability of nutrients has a marked effect on invertebrate densities, with low numbers occurring on nutrient poor heathlands compared with base-rich grasslands. Thus, because nutrients are released during fires, it might be expected that burning may cause at least a temporary increase in invertebrate densities after recolonisation of the burnt area. But it is also suggested in Section 4.2 that in the long-term, burning may cause a gradual decline in nutrient availability. Serial burning management may therefore contribute to the low densities of invertebrates on heathlands and may influence species composition by favouring species adapted to low nutrient conditions.

Secondly, invertebrate diversity is strongly related to soil and vegetation diversity. This has been well documented in the better known invertebrate communities of lowland heathlands (Webb 1986). In lowland heaths different species groups are favoured at different stages of vegetation development, and therefore variety in vegetation structure and composition is likely to be important in maintaining invertebrate diversity. However, the relatively few species which are unique to heathland tend to be most associated with the later stages of heath development. Burning in upland heaths may therefore increase invertebrate diversity if carried out on small scales that create fine grained habitat diversity. Furthermore, on upland *Calluna* heathlands, the greatest diversity of invertebrates is associated with stands in pioneer and degenerate phases (Barclay-Estrup 1974, Gimingham 1985), though caution needs to be given to interpreting these studies as they were carried out in an uneven aged area of heathland with the 'phases' occurring intermixed at very small spatial scales. Usher (1992) suggested that the high invertebrate diversity of heathlands may result from the periodic disturbance provided by rotational burning management. But, burning management aims to burn *Calluna* before it reaches its degenerate phase. Therefore, degenerate phase stands of *Calluna* are now rare in England (M. Rebane pers comm 2003), and thus the component of the upland heathland invertebrate community that is associated with this type of vegetation is likely to be missing in many areas.

Unfortunately it is difficult to test these hypotheses as very few studies have been carried out on the impacts of burning on upland invertebrates, and most of these have only investigated heather moorland communities. Furthermore, many of these studies provide simple descriptive snapshot comparisons of invertebrates numbers, diversity or species compositions on burnt and unburnt habitats. It is consequently, difficult to draw general conclusions on the underlying ecological relationships and overall impacts behind these observations.

6.1.2 Soil fauna

There is some evidence that soil microbia and fauna are affected by burning. For example, microbial numbers have been shown to increase after burning on shallow and deep peats, probably as a result of increases in pH and available nutrients (Maltby and Edwards 1984). However, in contrast, substantial reductions were observed in the densities of micro- and meso-fauna one year after a controlled burn of mature *Calluna* (Brown 1986). Total densities were much higher in the mature heather than in the recently burnt ground, being four and ten times greater in summer and winter respectively at 0 - 5 cm depth, and twice as high at 5 - 10 cm depth. Groups which showed large reductions in the burnt area were principally the litter-dwelling *Collembola* (springtails) and *Enchytraeid* worms.

Another study examined the longer-term response of soil fauna to an uncontrolled fire (Aked 1984). This found that eight years after the fire, soil fauna densities and diversities were lower in the burnt areas than the unburnt plots, and concluded that the intense burn, which led to the loss of litter, soil structure, fungal mycelia and vegetation cover clearly depressed soil fauna well below the level of normal management burns.

6.1.3 Macro-invertebrates

More studies have been carried out on the impacts of fire on macro-invertebrates utilising the soil surface and vegetation of upland habitats. These have tended to show that there is a succession in invertebrate colonisation after burning and that burning can promote invertebrate community diversity, particularly where patch sizes are small and recolonisation is not impeded. However, some species appear to be only associated with tall mature and degenerate stands, and may therefore only occur where there is no recent history of burning or very long burning rotations. Thus the studies may have been constrained in their abilities to examine the impacts of burning across the full range of potential invertebrate communities.

In a study of insect abundance on burnt and cut upland *Calluna* heath, Gardner and Usher (1989) found that considerable numbers of insects were present on the newly burnt and cut stands. Predatory and scavenging species such as Carabidae (ground beetles) and Formicidae (ants) did not appear to be adversely affected by the presence of bare ground between the *Calluna* stands. Similar results were also found in an earlier study of arachnids (spiders), where the hunting spiders *Gnaphosa leporina* and *Pardosa palustris* were most abundant along the margins of burnt and cut heathland, but otherwise showed little change in abundance from the original *Calluna* stand (Usher and Smart 1988). Although the loss of *Calluna* habitat may have been detrimental to phytophagous insects in the short term, there was also some evidence that the burnt areas (which were up to 35 m across) did not act as a barrier to their dispersal and redistribution.

Usher and Gardner therefore concluded that the short term effects of heather burning are not detrimental to the upland insect community. Furthermore, they suggest that the presence of significant numbers of insects on the open ground may provide good foraging opportunities for predators such as carabids, shrews, golden plover and red grouse with young broods. These results and conclusions should, however, be treated with some caution as both studies (Usher and Smart 1988, Gardner and Usher 1989) were based on pitfall trapping and catches therefore reflect activity levels as well as abundance.

In a later study of arthropods on *Calluna* heathland, Usher (1992) found that arachnid and coleopteran assemblages were primarily influenced by the *Calluna* growth phase. Several nationally rare species were also associated with the burnt or recently cut areas. Usher concluded that despite the low plant diversity and overwhelming dominance by one species (*Calluna*), heathland is horizontally and vertically structurally complex. This creates a wide diversity of habitats for insects, and traditional burning management ensures that these habitats are all present on the heathland at any one time. Thus "the habitat patches move around the heathland in time, but the continued existence of all patch types ensures the conservation of several associations of arthropod species". Merrett (1976) also came to the conclusion that traditional burning benefited spiders and other ground living invertebrates on lowland heaths.

However, it should be pointed out that these studies overlook the fact that late stands of mature and degenerate phases of *Calluna* that have not been burnt in recent times are now rare in upland moorland, and therefore their associated invertebrate assemblage is generally absent under traditional moorland management. It therefore seems that moorland beetles and spiders would benefit most from a twin approach of carefully managed prescribed burns and the permanent cessation of burning in some areas.

The occurrence of Lepidoptera has been studied in stands of rotationally burnt *Calluna-Vaccinium* heaths in Durham, Northumberland and southern Scotland (Haysom 1994, Haysom and Coulson 1998). This study assessed Lepidoptera abundance at the larval stage and related this by multiple regression to *Calluna* age, height, cover, green shoot density, and flower density. For most study areas and species, *Calluna* height explained most variation in larval density. Furthermore, after compensating for the effects of different study locations, it was found that the slope of the regression between the logarithm of the larval density and the logarithm of *Calluna* height was common to the macrolepidoptera, microlepidoptera, geometrid, and noctuid larval groups.

Larval diversity also progressively increased with *Calluna* height, due to the presence of uncommon moth species in the samples from taller *Calluna*, and a change in the contribution of common species to the community in different height zones.

On the basis of this study, and other relevant information, MacDonald and Haysom (1997) conclude that in the short-term burning destroys much of the food and shelter used by Lepidoptera and directly kills caterpillars and eggs of some species. However, in the long-term carefully controlled moor burning is beneficial if some stands are left unburnt. Provided that the burning rotations are sufficiently long (10-15 years) burning will create a diverse mix of stand ages and structures, and help maintain the presence of beneficial plants other than *Calluna*, such as *Vaccinium myrtillus*. As the diversity and density of the Lepidotera fauna is also related to *Calluna* height, it is also recommended that a substantial proportion of tall heather, ie over 20 cm, is maintained, preferably with some over 30 cm.

It would be unwise to extrapolate the results of studies of impacts of burning dry heaths on invertebrates to bogs. There are significant differences between bogs and dry heaths that have important influences on invertebrates, in particular, the structural variation of bogs is inherently greater than on dry heather moorlands with long management histories (Coulson *et al* 1992). Fewer studies have been carried out of the impacts of burning on invertebrates of bogs. However, a study of accidental fires in the Peak District has examined impacts on moorland which included *Eriophorum* dominated blanket bog (Anderson 1986). Although different species groups were found to react differently, marked differences occurred between the invertebrate fauna of unburnt and burnt areas. The burnt areas of the moor were found to hold higher numbers of Coleoptera and Opiliones (harvest spiders), but fewer Arachnids and other invertebrates. Differences were found to be less pronounced after two years and former numbers were restored after seven to ten years.

Some detailed studies have been carried out on the habitat requirements and influence of burning on the large heath butterfly *Coenonympha tullia*. This is a species of oligotrophic mires whose larvae feed on *Eriophorum vaginatum* and overwinter in its tussocks. These studies found that the presence of *C. tullia* was weakly positively correlated with burning, but weakly negatively correlated with severe burning (Dennis and Eales 1997, 1999). This is likely to be because *Eriophorum* spp tend to be favoured by moderately frequent burning (Hobbs 1984, see Section 5.7). Limited burning may also help by countering succession on drier sites and by encouraging *Erica tetralix*. The best conditions for *C. tullia* seem to be on sheltered mires, typically with peat that is more than 0.5 m deep, where dense, vigorous, overlapping growth of *Eriophorum vaginatum* and *Erica tetralix* occurs (Eales and Dennis 1998).

6.2 Amphibians and reptiles

Upland habitats hold few species of amphibian and reptile: only the common frog, palmate newt and viviparous lizard being widespread (Coulson *et al* 1992). No scientific studies were found by this review that examine the impacts of burning on any amphibians or reptiles. Although it is likely that some direct mortality will occur as a result of fires, the proportions of populations that are affected and long-term impacts are unknown. It is also possible that amphibians may be indirectly affected by erosion impacts which may lead to high levels of siltation in waterbodies. It can probably be assumed that detrimental impacts would occur if too frequent burning, or other inappropriate burning practices, lead to long-term changes in vegetation, particularly if this results in loss of dwarf shrubs and replacement by grasslands.

6.3 Birds

6.3.1 Red grouse

As described in Section 2.2.2, the aim of burning in many areas of heather moorland is the maximisation of grouse densities for shooting purposes. A brief review of the key effects of burning on red grouse is therefore provided below. However, the species has been subject to a huge amount of research into its ecology and management, and it is beyond the scope of this review to assess this in detail. Further information on red grouse ecology and management can be found in a number of research study reports and other reviews (Hudson

1986, Hudson 1992, Hudson and Newborn 1995, Redpath and Thirgood 1997, Smith *et al* 2000).

Grouse moor management traditionally entails the rotational burning of small strips of vegetation to create a patchwork of *Calluna* stands at different height and age stands (Miller 1980, Hudson and Newborn 1995). In particular the maintenance of healthy heather stands is critical for red grouse because *Calluna* is their principal food, it provides their only substantial shelter from predators and bad weather and provides cover for nest sites (Jenkins *et al* 1964, Moss 1972, Savoury 1978, Palmer and Bacon 2001). Traditional grouse moor burning management has been refined over the last two centuries (see Section 2.2) and has been shown to be effective in terms of increasing grouse numbers, at least in the short-term. For example, a study found that the average number of grouse shot on estates in northeast Scotland was correlated with the density of burnt patches, and furthermore, the local breeding population increased as a result of burning a large number of small patches over a period of four years (Picozzi 1968). A later regression analysis study showed that grouse bag density was positively correlated to a heather mosaic index, which is increased by burning, as well as the density of gamekeepers (which probably reflects the level of predator control), June temperature and heather productivity (Hudson 1992).

There are several possible reasons why red grouse numbers may be related to burning (Palmer and Bacon 2001):

- the general quality of the heather across the moor may be improved by the recycling of nutrients and/or the reduction of levels of secondary plant compounds;
- specific habitat requirements, such as heather stands of a particular age, quality or height, may be created at a higher density;
- the mosaic of smaller stands of differently aged heather may frequently provide two or more key habitat requirements that are close together; and
- the mosaic may provide certain factors, such as prominent edges between stands, which male red grouse use as references when establishing territory boundaries.

However, it should be note that the last three of these benefits may also be achieved, at least in part, by other management practices.

6.3.2 Community impacts

Two recent studies have examined the relationship between grouse moor management and bird populations. Smith *et al* (2001) studied the habitat characteristics of managed grouse moors, with the primary purpose of establishing whether changes in vegetation could alter the ratio of meadow pipits, and thus hen harriers (which feed on meadow pipits) to red grouse. However, they also examined effects on other bird species.

Grouse abundance and habitat was assessed on 69 1-km^{-2} sites in upland areas of northern England and Scotland where there was predator control and moor burning. Similar data were collected on 73 25-ha sites on the Langholm estate in south-west Scotland. This enabled the study to make a within-estate and among-moor comparison.

The results indicated that, in both the within-estate and among-moor assessments, meadow pipit abundance declined with increasing moor burning and heather. In the among-moor comparison, meadow pipit numbers increased with grass cover. Furthermore, linear regression analysis indicated that *Calluna* and *Sphagnum* cover and the area of moor burning could account for 42% of the variation in meadow pipit abundance. Furthermore, burning explained most of the variation (19%).

Analysis of the number of bird species and bird diversity was carried out on the among-moor data set. This indicated that the number of bird species declined with increasing *Calluna* and *Sphagnum* cover and habitat patchiness. Bird numbers were positively correlated with moor burning, but the effect was not quite significant. Bird species diversity (as measured by Simpson's index) significantly increased with higher levels of moor burning, but this only explained 9% of the variation. Diversity also increased from west to east, and this explained more of the variation (22%).

The study did not attempt to identify possible causes for these observed patterns in either meadow pipits, bird species richness or diversity. But Smith *et al* (2001) suggested that the observed negative relationship between moor burning and meadow pipits may be due to a reduction in invertebrate diversity as a result of burning, as well as a decrease in habitat structure. But this suggestion is not supported by the results of the invertebrate studies reviewed above (see Section 6.1).

The second study aimed to establish whether population densities of 11 species of breeding birds differed between heather-dominated moorland managed for red grouse shooting and other moorland with similar vegetation (Tharme *et al* 2001). This entailed a broad survey of breeding birds, vegetation and moorland management on 320 1-km squares on 122 estates in upland areas of eastern Scotland and northern England where grouse shooting is a widespread land use.

As would be expected the study discovered that rotational burning was more common on grouse moors than on other moors (34% more by area) and grouse moors tended to have less tall *Calluna*. Densities of breeding golden plover and lapwing were five times higher and those of red grouse and curlew twice as high on grouse moors as on other moors, while meadow pipit, skylark, whinchat and carrion/hooded crow were 1.5, 2.3, 3.9 and 3.1 times less abundant, respectively, on grouse moors. Although there were significant variations between regions, the differences in density between moorland types remained significant for golden plover and crow when these were taken into account. The differences also approached significance for lapwing and meadow pipit.

Poisson regression models were used to relate bird density to vegetation cover, topography, climate and soil type. Correlations of adjusted bird density were then made with measures of different aspects of grouse management. This indicated that there was a possible positive influence of predator control (assessed using crow density) on red grouse, golden plover and lapwing, and a positive effect of heather burning on the density of red grouse and golden plover. Burning had a negative effect on meadow pipit. This concurs with the results of Smith *et al* (2001) but not those of Hudson (1992), who found a positive relationship between meadow pipit numbers and grouse bags, but not keeper densities. There was also a suggestion that burning had positive effects on densities of black grouse, curlew, lapwing and whinchat. To some extent the results of the study resemble those of Thompson et al (1997). But it should be noted that lapwing is relatively scarce on heather moors, and therefore the impacts of heather management on its populations are unlikely to be significant compared with management of other habitats.

Taken together, these studies suggest that burning benefits some species, and slightly increases bird species-richness and diversity. But there seems to be agreement that it has a negative impact on meadow pipits. However, these studies have probably focused on a fairly restricted range of mainly dry *Calluna* dominated heathland habitats. Therefore, care should be taken in extrapolating these results to other habitats such as bogs, which as described before are likely to have a higher inherent structural diversity. Furthermore, detailed autecological research needs to be done to confirm these essentially correlative findings and identify biological causes.

No studies appear to have been carried out on the impacts of grassland burning on upland bird communities.

6.3.3 Species of conservation importance

Many of the birds of the British uplands are part of an internationally important assemblage of breeding species (Ratcliffe and Thompson 1995, Thompson *et al* 1995b). Although many of these are concentrated in Scotland, a number also have significant populations or important outlying range extensions in the English uplands. For example, some populations are the southernmost in the Western Palaearctic (eg merlin on Exmoor), or the world (eg golden plover, dunlin and red grouse on Dartmoor, and dotterel in northern England) (Brown and Grice 1993). Some are also listed on Annex I of the Wild Birds Directive.

Birds of conservation importance in the English uplands have been identified by English Nature (Brown and Grice 1993) and these are listed in Table 6.1. However, the English Nature list predates the UK BAP process and the latest assessment of the conservation status of birds in the UK (Gregory *et al* 2002). The table has therefore been modified accordingly and now indicates the status of each species with respect to these developments. Skylark has also been added as this is now a UK BAP Priority Species.

Birds may be directly impacted by burning, as a result of the potential destruction of nests, eggs and chicks during fires, or indirectly through impacts on habitats and other species (eg food resources, competitors and predators).

No studies appear to have been published on the losses of bird nests during prescribed burning, despite the fact that the last legal burning date of 15th April is after the beginning of the breeding season for several species. Furthermore species may also be impacted by licensed burning beyond 15th April, although as described in Section 2.2.3 this practice appears to be very uncommon.

Data on the breeding periods of birds in the uplands do not appear to have been published, and they are likely to differ significantly from general accounts of breeding seasons. This review has therefore carried out a preliminary analysis of egg laying dates of selected upland species in relation to the legal period for spring burning by using data from the Nest Record Scheme of the British Trust for Ornithology (H. Crick, unpublished data, 2003). These are presented in Table 6.2 and indicate the years from which the data derive, the number of Nest Record Cards in the sample, the earliest recorded date in the sample of the laying of the first egg in a clutch and a selected appropriate percentile of 1st egg dates. Thus, for example, 5% of all hen harriers in the UK have laid their 1st egg by 21st April. Where there are sufficient data then the information is provided on a country basis. The selected species are important upland species in England (Table 6.1) and characteristic upland species (ie meadow pipit) for which there are sufficient Nest Record Card data to carry out an analysis. Species that nest in trees or cliffs and buildings etc are excluded.

Table 6.1 Important birds of uplands in England (updated from Brown and Grice 1993)

Species	EN Priority*[1]	Wild Birds Directive Annex 1	Wildlife and Countryside Act Schedule 1	UK BAP Priority	Conservation status*[2]
Wigeon	High (2)				Amber
Teal	High (2)				Amber
Hen harrier	High (2)	*	*		RED
Buzzard	Medium				Greeen
Golden eagle	High (2)	*	*		Amber
Merlin	High (2)	*	*		Amber
Peregrine	High (2)	*	*		Amber
Red grouse	High (2)				Amber
Black grouse	High (2)			*	RED
Dotterel	High (2)	*	*		Amber
Golden plover	High (2)	*			Green
Lapwing	High (2)				Amber
Dunlin	High (2)				Amber
Snipe	Medium				Amber
Curlew	High (2)				Amber
Redshank	High (2)				Amber
Short-eared owl	Medium	*			Amber
Skylark	-			*	RED
Dipper	Medium				Green
Whinchat	Medium				Green
Stonechat	Medium				Amber
Wheatear	Medium				Green
Ring ouzel	Medium				RED
Raven	Medium				Green
Twite	High (2)				RED

Notes: *[1] Brown and Grice (1993); *[2] Gregory et al (2002)

Table 6.2 First egg laying dates for selected upland birds (source British Trust for Ornithology unpublished Nest Record Scheme data, 2003)

Species	Country	Years	Sample size	Earliest date	Percentile	Percentile date
Hen harrier	UK	68-99	117	11-Apr	5%	21-Apr
Merlin	UK	66-99	284	04-Apr	5%	26-Apr
Golden plover	UK	66-00	140	28-Feb	25%	17-Apr
Lapwing	Eng	62-98	635	16-Mar	50%	12-Apr
	Scot	62-98	119	26-Mar	25%	10-Apr
	Wales	62-98	64	21-Mar	25%	08-Apr
Dunlin	UK	66-99	109	30-Mar	5%	03-May
Snipe	UK	66-00	222	14-Mar	25%	17-Apr
Curlew	UK	66-00	218	10-Apr	5%	19-Apr
Redshank	UK	66-00	317	01-Apr	10%	18-Apr
Short-eared owl	UK	66-00	22	05-Apr	50%	01-May
Skylark	Eng	62-98	365	24-Mar	5%	17-Apr
	Scot	62-98	120	03-Apr	5%	19-Apr
Whinchat	Eng	62-98	295	06-May	25%	21-May
	Scot	62-98	323	10-May	25%	22-May
	Wales	62-98	330	09-May	25%	19-May
Wheatear	UK	66-99	506	14-Apr	10%	02-May
Ring ouzel	UK	66-97	837	12-Apr	10%	22-Apr
Meadow pipit	Eng	62-98	199	20-Mar	5%	17-Apr
Meadow pipit	Scot	62-98	377	17-Apr	25%	05-May
Twite	UK	66-99	334	30-Apr	25%	20-May

These data suggest that burning could jeopardise the nests and eggs of a significant proportion of golden plover, snipe, lapwing and redshank if burning is carried out in their nesting habitat. Some clutches of hen harrier, skylark, meadow pipit, merlin, curlew and ring ouzel may also be at risk. But it is unlikely to be causing a problem for whinchat, wheatear and twite. There are too few data to be able to assess impacts on short-eared owls reliably.

However, this analysis should be regarded as a preliminary assessment as egg laying data for some species (especially lapwing and redshank) include clutches from lowland breeding birds and these are likely to breed earlier than upland birds. It would also be worthwhile further screening the Nest Record Card data according to habitat type and sub-dividing the analysis by region where there are sufficient data to do so.

Nevertheless, the data make a strong case for further urgent research into the issue. Particularly as climate change may exacerbate the overlap between nesting and burning. Evidence from the Nest Record Scheme has already indicated that breeding has advanced by at least one week as a result of climate change (Crick *et al* 1997, Crick and Sparks 1999).

Few scientific studies have been carried out on the indirect impacts of burning on birds in the uplands. Nevertheless it is possible to predict to some extent the likely magnitude of impacts and key factors that may influence them from the studies reviewed above and general literature on their autecology. A summary of these are provided in Table 6.3.

Table 6.3 A summary of predicted impacts of upland burning management on some birds of conservation concern in England

Species	Impacts	References
Hen harrier	Detrimental if some tall heather is not retained for nesting, also highest numbers occur where meadow pipits and voles are abundant, and these favour grassland habitats. There appears to be a negative relationship between burning and meadow pipits densities (see above).	(Redpath *et al* 1998, Smith *et al* 2001, Tharme *et al* 2001, Redpath *et al* 2002)
Golden eagle	Extremely rare in England and effects of burning are unknown. The species favours areas with high livestock and carrion densities.	
Merlin	Detrimental if some tall heather is not retained for nesting where tree nest sites are absent. Densities are correlated with meadow pipit densities, so burning may be disadvantageous.	(Coulson *et al* 1992)
Peregrine	Probably no significant impacts as the species takes a wide variety of birds and nests on cliffs.	
Black grouse	Some evidence that burning is beneficial on heather moorland. But the species favours habitat mosaics with scrub / scattered trees which are not encouraged by burning. Also detrimental if some tall heather is not retained for feeding, roosting and nesting.	(Coulson *et al* 1992, Tharme *et al* 2001)
Golden plover	Some evidence that burning is beneficial on heather moorland as the species favours short-vegetation in young heather stands (<12 cm high), but may also be related to keeper densities and predator control.	(Coulson *et al* 1992, Hudson 1992, Tharme *et al* 2001)
Short-eared owl	Little information, but probably detrimental if some tall heather is not retained for nesting. Also frequent burning of heather and grasslands likely to be detrimental as the species requires areas with high densities of accessible voles.	
Skylark	No scientific data, but may benefit from frequent burning which encourages grassland rather than dwarf shrubs.	
Ring ouzel	Unknown	
Twite	Little information, but burning *Molina* may be beneficial eg in the southern Pennines, where the spring burning exposes a rich sources of seeds	(Backshall *et al* 2001)

6.4 Mammals

6.4.1 Mountain hare

The mountain hare is the only mammal that is truly characteristic of upland habitats in Britain, and the only mammal of conservation importance in such habitats. In Britain it is most numerous in north-east Scotland (where most research has been carried out on it), where it appears to benefit from moorland management for grouse. In England, it only occurs in the Pennine area of south Yorkshire and Derbyshire where it was introduced in about 1880 (Corbett and Harris 1991).

The mountain hare usually occurs at low densities, typically <0.1 ha-1 (Hobbs and Gimingham 1987). *Calluna* is the most important part of the diet, constituting 90% of intake in winter, but decreasing to 48% in summer (Hewson 1962). Numbers appear to be related to nutrient levels in the soil and associated levels of nutrient availability in *Calluna* shoots (Gimingham *et al* 1979). In areas with few hares, their feeding preference appears to be for young (pioneer) heather, but where densities are higher then they switch from a preference for pioneer heather to building and mature heather in winter and spring (Hewson 1976). However, if pioneer heather patches have more than 2% cover of grass then it is less favoured than early building heather (Hewson 1989).

No studies were identified by this review that directly examined the impacts of burning on mountain hares. Nevertheless, it can surmised from the available information on their habitat and diet preferences that the species is likely to benefit from carefully managed traditional heather moorland burning practices that maintain a significant proportion of *Calluna* in early growth phases. Indeed, an investigation of the relationship between mountain hare numbers and grouse shot on estates in north-east Scotland revealed that 59% of grouse moors (of 37) showed a positive association between the number of hares shot and grouse numbers (Hudson 1992). As the number of hares shot probably reflects hare density this suggests that hares benefit from the same management practices that benefit grouse. But, although such beneficial management practices may include burning, Hudson states that it is likely that the association is due to more effective fox control measures on estates with high grouse numbers.

It can also be predicted that frequent burning, which increases the proportions of grass present would be detrimental. Additionally, hares tend to shelter from snow and bad weather in old tall stands of *Calluna*. Therefore, burning may be detrimental if too high a proportion of the moor is under short rotation burning management. Overall, therefore, mountain hares are likely to benefit most from a twin-track strategy of moderately frequent burning, to maintain short feeding areas, plus scattered fire-free areas to provide tall vegetation and some scrub for shelter and winter forage. Fire free areas may also allow woodland establishment, which may provide further benefits as mountain hares will use these for winter browsing (Hulbert *et al* 1996).

6.4.2 Mammals as food resources for raptors

A number of mammals are of importance as food sources for predator species of conservation importance, and therefore the impacts of burning on their populations are briefly considered here.

The principal species of value to upland predators of conservation importance is the short-tailed field vole, which is an important prey for buzzard, kestrel, short-eared owl and hen harrier, as well as a number of mammals including foxes, stoats, weasels, polecats, pine martens, wildcat and badger (Coulson *et al* 1992). No studies of the impacts of burning on vole populations in the uplands were identified in this review, but impacts may be predicted from their known habitat requirements. In general the species requires rough, ungrazed grassland, such as rank *Molinia* and *Juncus* communities, whilst numbers on blanket bogs are low (Corbett and Harris 1991, Coulson *et al* 1992). The species is therefore likely to benefit from burning practices (such as frequent burning) that encourage the spread and dominance of grass species at the expense of *Calluna* and other dwarf shrubs.

The other mammal that is likely to be a common prey for upland predators is the rabbit. This is likely to be important prey for the buzzard as well as foxes and stoats. Rabbit populations, however, fluctuate considerably in the uplands as a result of myxomatosis and recovery is sporadic. Again there appear to be no studies on the impacts of burning, but it is probable that the species will also benefit from burning practices that favour grasses. On heaths disappearance of rabbits appears to be associated with increases in the cover of dwarf shrubs, especially *Calluna* (Coulson *et al* 1992).

7. Conclusions

7.1 Current burning practices in the English uplands

- Prescribed burning remains a common practice in much of upland England (and elsewhere in Britain) for agricultural purposes or grouse moor management, or a combination of both. One of the key aims of burning is the maintenance of open grassland or heathland habitats by the prevention of succession and establishment of scrub (eg gorse) and woodland. Prescribed burning for agricultural benefits also aims to improve the quality of forage for livestock by removing old vegetation, which releases nutrients and stimulates vigorous, palatable and nutritious new plant growth. Removal of litter encourages fresh grass growth in spring (spring bite) and increases accessibility to livestock, whilst burning can also rejuvenate *Calluna* and other evergreen shrubs, which are particularly important sources of winter forage. Burning also aims to limit the height of the vegetation to allow access to sheep, although some taller areas may be retained to provide shelter and accessible forage during periods of snow.

- Similarly, burning on grouse moors aims to provide palatable and nutritious vegetation for livestock and red grouse, primarily by rejuvenating and maintaining *Calluna* in its more productive growth phases. It also aims to provide a small-scale patchwork of heather stands in different growth stages, to provide good feeding areas close to taller heather, which provides shelter, nest sites and cover from predators.

- Despite its common use and obvious impacts on uplands, there appear to be few statistics available on current burning practices in England. In most upland areas the number and location of fires are not recorded and little information exists on their actual size or frequency. Consequently it is not possible to quantify the proportion or frequency with which different uplands areas are burnt in England. However, it is well known that burning management remains a common practice on the grouse moors of Northumberland, the Pennines, the Forest of Bowland and the North York Moors. Many upland rough grasslands such as on Dartmoor, Exmoor and Lake District are regularly burnt for agricultural purposes.

- Purely agricultural and grouse moor management burning practices differ considerably. Agricultural burns are usually uncontrolled and therefore tend to cover very large areas in one burn, typically from tens to hundreds of hectares. Grassland burning rotations are often very short, with fires typically returning every couple of years or annually in some areas.

Prescribed burns for grouse moor management are usually planned and controlled, and carried out by burning strips of only about 30 m width (and typically about 0.5 ha in total area). Where *Calluna* dominates, the normal aim is to determine the length of burning rotation according to the time it takes for heather to reach the appropriate stand conditions for burning, which is judged by height, rather than age. Generally it is recommended that burning is carried out when most *Calluna* reaches a height of 20 – 30 cm in pure stands. This should typically result in a fire return period of about 10 – 15 years, in which case 1/10th to 1/15th of the land would be burnt on average each year.

Information on the actual fire frequencies achieved on grouse moors in England is lacking, but some evidence suggests that there has been a decline in the practice of rotational burning in recent decades on the North York Moors, where a high proportion of heather was found to be over 40 cm and in its mature or degenerate growth phases. But it is not clear whether this is typical of moorland, as there is also widespread concern in English Nature, based on observations by field staff, that burning management has intensified on some grouse moors, including the frequency of burning rotations.

7.2 Generic impacts

7.2.1 Soils

Most well controlled prescribed fires have little direct effect on soils, because litter, humus and the soil itself are good insulators, particularly when moist. However, there are usually significant indirect effects, including changes to the soil micro-climate as a result of the loss of vegetation cover, reduced peat formation, reduced water infiltration rates, increased water retaining capacity and increases in pH. Most importantly, erosion rates increase as a result of the exposure of bare soil to rainfall, wind and frost action, which results in a net loss of soil material until vegetation cover is regained (which takes about 15-20 months). Although there is some uncertainty over the actual quantities of soil lost, most studies indicate that erosion rates are probably higher on large burnt areas and on slopes. It has also been found that erosion is 10 times greater in winter than in summer.

Very hot fires may result in direct physical damage to the soil, through combustion of organic humus layers and if peat is present, this may ignite under dry conditions. Peat layers may then burn for long periods leading to substantial soil loss and severe long term effects, with prolonged periods of bare ground and erosion. Burning and drying of adjacent peat layers also results in the release of stored carbon and thus emission of carbon dioxide and other 'greenhouse' gases. The prevention of fires that ignite humus layers and peat soils must therefore be one of the highest priorities of fire management policies.

As with many other issues concerning burning, most studies of fire impacts on soils have only investigated the short-term impacts of single fires. Therefore much more research needs to be carried out on the cumulative and long-term impacts on soils of sequential management burns.

7.2.2 Nutrient dynamics

Burning of vegetation has a substantial effect on nutrient dynamics as it results in the partial release of nutrients. Some of these nutrients are re-incorporated into the soil and become available to plants, which stimulates plant growth and temporarily increases the nutritional value of the regrowing vegetation.

However, some of these nutrients are lost (or redistributed) in smoke (especially in hot fires) or through erosion of ash. But these losses may be replaced with time through inputs in rainfall. Various studies have therefore attempted to calculate the overall impacts of burning on nutrient budgets and generally concluded that most nutrients lost in prescribed heathland fires will be replaced by rainfall within the typical fire return period. But there maybe shortfalls in replacement of potassium, nitrogen and particularly phosphorus, and these deficiencies could be exacerbated by more frequent burning, severe fire conditions and erosion. But the overall effects of long-term burning on nutrients are unclear as most studies have only looked at single burns, and the one study that examined the cumulative effects of serial burning found that burning increased, or at least depleted less, the mineral fertility of the soil in comparison with unburnt stands.

Most importantly the significance of any possible long-term decline in the nutrient status of moorlands as a result of burning is poorly understood. If nutrients are being lost, this could reduce the productivity and quality of the vegetation for livestock, grouse and other herbivores, with unknown knock-on impacts on biodiversity. But on the other hand, heathland ecosystems appear to be adapted to low nutrient conditions, and nutrient losses through burning may not be detrimental. In fact they may contribute to the maintenance of the habitat. And this may be especially important under current circumstances where atmospheric pollution has significantly increased nutrient inputs from rainfall beyond natural levels. Thus it is clear that much more research needs to be carried out on nutrient dynamics and requirements in upland ecosystems, particularly with respect to long-term cumulative impacts of burning.

7.2.3 Hydrology

As noted above, fires may result in a reduction in rates of water infiltration into soils, whilst the loss of vegetation cover will reduce rates of rainfall interception and evapotranspiration, all of which would be expected to lead to increased rates of runoff. And some studies have observed a reduction in the water storage capacity of the soil and increased peak flows following burning. But relatively few studies have been carried out on the hydrological impacts of burning at the plot or catchment scale, and those that have been carried out in British uplands have produced partially conflicting results. Further studies are therefore required to examine potential impacts of burning more thoroughly and under a range of habitat and environmental conditions.

7.2.4 Habitats, vegetation and plant species

Fire is a natural component of many ecosystems, and since the last ice age has probably occurred relatively frequently in upland habitats in Britain. As a result most of the principal upland species have some form of adaptation to surviving or regenerating from fires. Humans have also frequently used fire for vegetation management over much of the last post-glacial period, and this has further influenced the vegetation composition of upland habitats. Thus, although a high proportion of the English uplands were formerly tree covered, the vast majority are now open treeless habitats dominated by grasses, dwarf shrubs and in wetter locations bryophytes such as *Sphagnum* spp. Species that are either fire sensitive or have very slow recolonisation rates are now mostly absent from upland habitats that are subject to burning.

- Burning remains a common traditional management practice over much of the English uplands, but the extent to which different habitats are routinely burnt is not monitored or quantified. From general accounts it is possible to conclude that most upland heathlands (particularly drier heaths) are subject to burning; typically every 10-15 years for grouse management purposes. Many upland acid grasslands are regularly burnt for agricultural purposes. Most areas of remaining blanket bog in England are also under some form of burning management, despite guidance in the Heather and Grass Burning Code (MAFF 1994)that these habitats (ie peat) should not normally be burnt.

- But not all upland areas are affected by regular fires. Burning management appears to generally decline with altitude, and therefore montane heathlands and grasslands are probably seldom burnt, except perhaps accidentally. Wet habitats also act as fire refugia except during dry conditions, which are normally outside the burning period. Similarly most rocky habitats are unlikely to be able to sustain fires. Woodlands, calcareous grasslands and improved grasslands in the uplands are probably only subject to occasional accidental fires and are not intentionally burnt.

The impacts of prescribed burning on plants, vegetation communities and habitats is complex because (as described further below) it depends on the type and the effectiveness of each plants' adaptations to fire, the condition of the habitats before the fire, the characteristics of the fires, the cumulative impacts of past fires and interactions with grazing and weather etc. However, a summary of the most typical advantageous and detrimental impacts from prescribed burning management on habitats is provided in Table 7.1 (a summary of impacts on key plant species has been previously provided in Table 5.2).

Table 7.1 Summary of advantages and disadvantages of prescribed burning for nature conservation purposes

Broad Habitat	**Extent of burning**	**Advantages**	**Disadvantages**
Calcareous grassland	May be burnt accidentally and when part of a mosaic, extent and frequency unknown	Uncertain	Uncertain
Acid grassland (including grassland on heath and peatland habitats)	Large areas burnt frequently, particularly on the south-west moors	Occasional burning may prevent *Molinia* dominance.	Frequent fires encourage *Molinia* dominance and lead to loss of bog species and lead to further degradation of blanket bog habitats.
Bracken	Uncertain, probably occasional	Can be used to weaken plants and increase their susceptibility to other treatments (eg herbicides).	By itself it often encourages further bracken expansion if *Calluna* regrowth is poor.
Dwarf shrub heath	Extensive, particularly in grouse moor areas	Maintains open moorland by halting succession, increases structural diversity and species diversity if appropriately managed.	Encourages grasses and reduces species and structural diversity and if too frequently burnt, burning old stands risks long re-establishment period and erosion, which may be exacerbated by high fire temperatures. Risk of peat combustion on wet heaths in dry conditions. Loss of older stands of heather, removal of nest sites etc.

Broad Habitat	Extent of burning	Advantages	Disadvantages
Fen, marsh and swamp	Not affected if permanently wet, but seasonally wet habitats may be burnt accidentally	None	Loss of potentially fire sensitive species.
Bogs	Majority are probably under some form of burning rotation	Favours *Eriophorum*, which can benefit black grouse and large heath butterfly if *Eriophorum* abundance is low. Some carefully selected and controlled burning may be necessary for reducing risk of severe wildfires in high risk areas.	Potential loss of fire-sensitive species and tendency to become dominated by *Eriophorum* on short rotations, or *Calluna* on long rotations. Nutrient losses may be significant. Reduced peat formation and significant risk of erosion and combustion of peat. Peat drying and combustion causes significant carbon losses.
Montane habitats	Probably very rare	None	Potential complete loss of fire-sensitive species. Very high erosion risk.
Inland rock	Probably rare, usually a fire refugium	Control of scrub in some circumstances.	Potential complete loss of fire-sensitive species.

7.2.5 Invertebrates

The ecology of invertebrate communities in upland habitats has been relatively little studied. Nevertheless, there is sufficient information to indicate that the abundance of upland invertebrates is related to nutrient availability and overall productivity of the habitat, whilst invertebrate diversity is related to soil and vegetation diversity, with certain species being associated with particular phases of the *Calluna* growth cycle.

Few examinations have been carried out of the impacts of burning on upland invertebrates, but studies of dry heaths suggest that burning initially increases microbial activity (probably by releasing nutrients) and temporarily reduces the abundance of soil invertebrates. However, in the longer term well controlled rotational burning may be beneficial in terms of diversity (particularly if old heather is not burnt). Burning in small patches with 10-15 year rotations, as traditionally carried out on grouse moors, helps to increase diversity by creating a mosaic of *Calluna* stands with different growth phases, and therefore with different structures and associated species. Such patches are also small enough to enable colonisation and dispersal. But, under burning management, late mature and degenerate *Calluna* phases will not normally be present, and therefore invertebrate assemblages associated with these growth stages will be absent. Moorland invertebrates would therefore probably benefit most if some areas of habitat are permanently taken out burning management alongside patches under typical 10-15 year burning rotations.

The apparent benefits of burning heathlands probably do not apply to bogs, as these are more naturally diverse and contain an invertebrate fauna that is dependent on wetland components including the occurrence of certain lower plants and standing water. But insufficient studies have been carried out of burning on bogs to come to any conclusions on this subject.

7.2.6 Birds

Increasing the abundance of red grouse is one of the key aims of burning on many upland heaths. This aspect of burning management has been subject to several studies which have demonstrated that traditional prescribed burning (as well as other grouse moor management practices) increases grouse densities.

The effects of burning on other birds have been little studied, particularly in other habitats. But some recent large-scale correlative studies have examined the effects of burning and other grouse moor management practices on some other bird species. These studies concluded that burning on heathlands slightly increases species-richness and diversity. From analysis of a range of studies, burning impacts on individual species appear to be detrimental to meadow pipits, but advantageous to red grouse and golden plover, and possibly black grouse, curlew, lapwing and whinchat. However, most studies have focussed on a fairly restricted range of mainly dry heathland habitats, and therefore these results should not be extrapolated to other habitats, such as bogs. No studies have been carried out of the effects of large scale agricultural burning of grassland (and grass dominated degraded bog and heath habitats) on bird communities.

A preliminary analysis of first egg laying dates (from BTO Nest Record Card data) suggests that several upland species could lose nests and eggs if burning is carried out within their habitats towards the end of the burning season (15th April). Of the early breeding species those that are most likely to be at risk include golden plover, meadow pipit and skylark, but other species may also be affected where bog and wet heaths are burnt. The overall impacts of such nest losses on breeding success and long-term populations is unclear, but these could be exacerbated by climate change which appears to be resulting in early breeding in many birds.

7.2.7 Amphibians, reptiles and mammals

Upland habitats hold few species of amphibian and reptile, and none of these are of high conservation concern. No scientific studies have been found that have examined the effects of burning on any of these species. But it can be assumed that burning practices that result in significant habitats changes, such as loses of heather cover, could have substantial indirect effects on amphibians and reptiles.

Similarly, most mammals that occur within upland habitats are widespread and common. The mountain hare is the only truly characteristic mammal of the British uplands, and only occurs in England as a small introduced population. Research on mountain hares (mostly carried out in Scotland), suggests that the species benefits from grouse moor burning practices as it also mainly feeds on young nutritious *Calluna* shoots. However, the mountain hare may also benefit from setting aside some areas from burning management as tall heather provides shelter and winter forage.

The short-tailed vole, and to a lesser extent some other small mammals, are particularly important as food sources for some raptor species of particular conservation concern, including short-eared owl and hen harrier. Little information exists on the impacts of burning on these species, but it is known that most are reliant on grass rather than shrubs such as *Calluna*. Thus it can be reasonably reliably predicted that these species will tend to benefit from burning practices that favour tall rank grass over shrub cover, whilst retaining some litter cover.

7.3 Key factors affecting impacts

7.3.1 Frequency of burning

The pre-fire composition and age of heathland vegetation stands are the most important attributes that determine the outcome of fires in heathlands, and other upland habitats where *Calluna* is a significant component (see Section 5.1). These in turn are principally determined by the frequency of burning. The pre-fire composition of the stand is of critical importance as post-fire re-establishment will normally be by vegetative means. Thus if *Calluna* is the dominant species in the pre-fire community, then it will generally regain dominance eventually, provided that reasonably vigorous vegetative regeneration is able to take place.

However, even under ideal conditions it takes several years for *Calluna* to re-establish dominance, and in many situations, such as in wet heaths and bogs it will take 15-20 years. During the period of *Calluna* re-establishment other faster growing species such as grasses are able to establish temporarily, before they are shaded out by the *Calluna*. Thus, there is usually a sequence of plant establishment and dominance after a fire. But if heather is burnt too frequently the sequence is not completed and the slower growing shrubs are gradually lost and replaced by faster establishing species, such as *Nardus* and *Molinia*, or *Eriophorum* on bogs.

Consequently, it is likely that frequent burning on the south-west moors, where it is traditionally carried out every couple of years, and in some places annually has encouraged the spread of large expenses of *Molinia* dominated grassland at the expense of *Calluna* and the former blanket bog habitats. However, high grazing pressure may also be a contributing factor in many areas.

The result of burning old *Calluna* stands (in their mature and degenerate phase) is extremely slow regrowth of *Calluna*, as this is by seed germination rather than vegetative means, and in some cases persistence of bare ground for many years after the fire. In other situations *Calluna* may be out-competed by *Vaccinium* and *Pteridium* which may invade and attain long-term dominance.

The frequency with which heather should be burnt depends very much on local conditions and specific management objectives. In particular, it depends on the growth rate of *Calluna*, which varies according to climate, altitude and soil conditions etc. It is therefore generally recommended that the optimal interval between burning should be determined by the height and growth phase of the stand, rather than a specified interval. Traditional heather moorland burning is usually carried out when *Calluna* reaches 20-30 cm in height, ie in the late building phase. Typically this results in a rotation length of 10-15 years.

For nature conservation purposes it is beneficial to provide a wide variety of heather ages, growth phases and heights, including heather that is taller than 30 cm and in the mature and degenerate growth phases. However, it would be unwise to attempt to provide older and taller heather by merely lengthening the burning rotation, because as described above, the vegetative regeneration capacity of mature and degenerate heather is low. Tall stands also produce hot fires, which may further reduce the capacity for vegetative regrowth (see below). It will therefore be necessary to exclude stands of old and tall heather from the burning rotation.

7.3.2 Intensity of burning

Fire temperature and ground surface intensity are also key factors that determine the impacts of a fire. This is because many moorland species, such as *Calluna* may be killed by temperatures attained in some fires, despite their adaptations to fires. Well controlled management fires of heathland stands of 20 – 30 cm height are unlikely to have significant effects on the regeneration capacities of most species. But older, taller stands create hot fires that may kill off buried shoots and buds etc, particularly in dry conditions, on shallow soils and where there is no overlying protective moss cover. This may substantially reduce the potential for vegetative regeneration. Hot fires also lead to increased losses of nutrients in the smoke, which may lead to nutrient depletion in the ecosystem, but more research is needed to confirm this.

In dry conditions there is a high risk that hot fires will ignite peat and humus layers, which can then smoulder for extended periods. A smouldering fire on the ground surface has the greatest and most damaging effect of all on the vegetation and soil because of the long duration of lethal temperatures. This can lead to the loss of the seedbank and overlying organic soils. Vegetation re-establishment is then likely to be very slow (sometimes taking decades), which further encourages prolonged periods of erosion.

7.3.3 Size of burn areas

In general burning should be carried out in small patches. Wide burns tend to be hotter than narrow burns, at least in part because they create stronger convention currents, which tend to feed the fire. Consequently, they are also more difficult to control and, being hotter, may cause more damage to vegetation and soils. However, large fires also tend to be very patchy, with large variations in ground surface intensity, so some areas may not be effectively burnt. Large fires also lead to the increased risk of significant erosion impacts, partly due to the volume of ash and other material that may be eroded. But, large bare areas are also more likely to build up sufficient run-off during heavy rain for larger materials to be moved.

In general, biodiversity is promoted by increasing structural diversity. Therefore small fires that create a patchwork of vegetation stands of different ages and species compositions are likely to be most beneficial. Thus, the burning of strips of c. 30 m wide that are now typically advocated for grouse moor management is likely to contribute to habitat diversity, provided that the burns are appropriate with regard to frequency and are carefully controlled, and sensitive areas remain unburnt.

But in contrast the large, primarily agricultural, burns that are carried out on the south-west moors, and elsewhere, are likely to be contributing to a loss of structural and floristic diversity. Such large burns may also be encouraging the expansion of *Molinia*, as they tend to create a super abundance of fresh growth. This cannot be grazed off by the livestock present (see 7.3.5).

7.3.4 Timing of burning

Although the legal burning period in the uplands appears to be quire long (ie from 1st October to 15th April, though extendable under licence), the actual number of days when burning can take place in practice is limited by weather conditions and the short periods of daylight. As a result most prescribed management burns on heathlands and grasslands actually take place over a relatively short period in the spring when weather conditions are generally most suitable. Some burning also takes place during the autumn, but relatively little takes place during the mid-winter period.

Spring burning has some advantages over autumn and winter burning beyond the likelihood that weather conditions will be most suitable. Firstly, spring burns tend to be easier because the vegetation and litter usually has a lower moisture content. Thus burns tend to be more predictable and uniform. They also tend to be more effective in removing litter. This is an important aim of grassland management fires, and on heathlands such 'clean burns' produce the best seedbed, though with good burning practices this should not normally be required (in which case 'clean burns' are undesirable). If regeneration is likely to be by seedling germination, spring burning will result in germination at a time when seedlings will be less exposed to uprooting by needle-ice.

However, there is some evidence that there could be advantages from autumn burns, and lengthening the autumn burning season (eg allowing burning in September). Firstly, *Calluna* appears to re-establish better after autumn burns, probably because other competitive species are dormant over the winter.

Secondly, autumn burns also tend to produce cooler fires because of the typically higher moisture levels in the fuel and litter layers. This minimises potential impacts on mosses and lichens, and reduces the risk of damaging the regenerative capacity of *Calluna* and other shrubs. Cooler fires with damp litter and soils also reduce the risk of peat ignition and subsequent impacts. But any burning increases erosion as a result of the removal of vegetation cover, and autumn burning increases this further because burnt areas are more exposed to needle-ice and other extremes of winter weather. If litter layers are retained under cool fires this may give some protection, but nevertheless, it is unwise to carry out autumn burns on areas that are particularly vulnerable to erosion (eg on slopes).

Lastly, autumn burning avoids the risk of destroying the nests and clutches of breeding birds. And there is evidence (see Section 6.3.3) that spring burning could be a significant problem in this respect as the spring burning season (ending on 15th April) overlaps significantly with the start of the breeding season for several bird species of conservation importance.

7.3.5 Interactions with grazing and other management measures

Virtually all areas of upland vegetation that are burnt will be subject to pre- and post-fire grazing to some extent, and therefore the effects of burning and grazing are inextricably linked. The effects of grazing depends on the vegetation types affected, the fire return frequency, size and locations of burns, and the number and type of stock grazing the burnt areas and the seasonal timing of grazing. But these aspects of interactions between burning and grazing have been little researched and are not understood in detail.

In general grazing and burning impacts appear to affect upland vegetation in similar ways. For example, increased burning frequencies and grazing intensities on heathlands both tend to change vegetation from *Calluna* dominated communities to grasslands. Thus grazing tends to accentuate the impacts of burning. It has also been demonstrated that high grazing rates can have substantial impacts on post-fire heathland development by influencing the rate of *Calluna* recovery. Consequently it has been suggested that the substantial losses of heather moorland to grassland which have occurred in the uplands is the result of significant increases in grazing pressure combined with poor burning practices.

But when grazing is heavy regular management by burning may be particularly important to maintain heather dominance. Because grazing animals concentrate on areas of younger heather overgrazing of regenerating heather may occur if insufficient areas are burnt. Thus reductions in stocking density or shepherding may be required if areas of heathland are taken out of burning management.

On *Molinia* dominated grasslands (and former bogs or heaths that are dominated by *Molinia*) it is important to ensure that spring and summer grazing pressures are sufficiently high to remove all or a high proportion of the new growth following burning. This helps to check the spread of *Molinia* and avoids the build up of more litter. However, this does not appear to happen in many grassland areas, such as Dartmoor, where uncontrolled fires are routinely lit that burn larger areas than can be effectively grazed with the available livestock. As a result litter accumulation is rapid and burning is consequently undertaken on a regular basis, which in turn further exacerbates litter build up and *Molinia* dominance.

7.3.6 Methods of burning

Most fires are set to burn with the wind (head-fires) but back-fires, which are controlled to burn against the wind are sometimes used, eg to burn firebreaks, wet ground or old heather. No studies appear to have been undertaken specifically on burning methods. However, it is known that back-fires are slow moving, consume more fuel and release much of their heat close to the ground. They therefore tend to produce greater impacts on the ground surface, typically producing a 'clean burn' which removes all litter and is therefore most damaging to lichens and bryophytes etc. It may also be predicted that back-fires will increase the risk of killing off basal shoots etc, thereby reducing the potential for vegetative regeneration, which would be especially harmful where back-fires are used to burn old stands.

7.3.7 Slope, aspect and altitudes

Some studies have investigated the influence of slopes on post-fire erosion rates, and although these have tended to be inconclusive, it is generally considered that erosion increases in relation to slope. Slopes also tend to produce hotter and faster moving fires, which in turn increases the likelihood of such fires getting out of control. Hot and large fires further increase the risk of erosion. Thus, special care needs to be taken with burning on slopes, eg by burning small patches, avoiding gullies, eroded ground, and old *Calluna* stands. Slopes of 1:3 should not be burnt.

No specific information appears to be available on the effects of aspect and altitude on burning impacts. However, it is widely acknowledged that montane habitats are particularly sensitive to disturbance (eg due to slow vegetation growth rates, shallow soils and extreme climatic conditions), and should therefore not be burnt.

7.4 Recommendations

7.4.1 Generic recommendations for burning practices

It is not within the scope of this study to provide detailed recommendations for future burning practices. In part this is because such recommendations would need to take into account considerations other than nature conservation, such as wildfire prevention and control measures, health and safety issues, landscape objectives, archaeology protection needs and practical constraints, such as possible labour shortages. Nevertheless, from the conclusions of this scientific review it is possible to provide some generic recommendations that are based on conservation objectives.

It is evident from this, and other reviews (eg Hobbs and Gimingham 1987, Mowforth and Sydes 1989, Coulson *et al* 1992, Shaw *et al* 1996, Backshall *et al* 2001), that carefully controlled burning practices can play an important role in the maintenance of some semi-natural upland habitats in England. Such environmentally sustainable burning management practices need to be promoted, with careful planning, control and variations to promote biodiversity. A general aim should be to encourage a diverse range of <u>appropriate</u> environmentally sustainable burning treatments to be used across a variety of temporal and spatial scales. Particular recommendations are as follows:

1. Because burning impacts vary considerably according to local circumstances and conservation objectives, it is recommended that burning plans are produced for all areas. On SSSIs and other designated sites of conservation interest, the plans should be agreed with English Nature and should take into account and contribute to the conservation

objectives of the site. Given the important interactions between post-fire grazing and vegetation recovery, it is essential that burning plans are integrated with livestock stocking rates and other management practices, which are adjusted if necessary. Burning plans should also be components of agri-environment scheme management agreements where appropriate.

2. Special efforts should be taken (eg by awareness raising, training and perhaps revised regulations) on all moors to minimise the risk of wildfires, particularly in areas of high biodiversity importance, such as bogs, species-rich heathland and grasslands, and areas with rare and characteristic upland species.
3. In common with other studies it is recommended that the following habitats should not be burnt:
   - Woodland, scrub and scattered trees, in particular where *Juniperus* is present.
   - Pteridium stands, unless in combination with other control measures.
   - Montane heathlands and grasslands.
   - Screes and other rocky habitats.
   - Blanket bogs and other mires should not be burnt unless there is a specific and clear overall conservation benefit, eg in high fires risk areas, a significant reduction in the risk of severe wildfires that may ignite peat layers. If burning must be undertaken, then it should it should be no more than once every 20 years, and should be carried out when the litter / *Sphagnum* layer is wet.
4. Areas within any habitat that are used as traditional nest sites by protected species (eg merlin and hen harrier) should not be burnt
5. Burning should be avoided on areas with bare peat or soil, shallow soils (< 5cm), steep slopes (> 1:3), along water courses and where erosion has occurred after previous fires.
6. Old stands of *Calluna* (ie late mature or degenerative phases) should not be burnt as these have low regeneration abilities and are prone to invasion by undesired species; they are also likely to produce hot or unpredictable fires which create further problems. Such stands should not be brought back into burning management unless there is a clear conservation reason which outweighs the potential risks. If burning is to be reinstated, it should if possible be preceded with a programme of cutting. Burning should be undertaken with great care and post-fire treatments (eg bracken control) provided if necessary.
7. Where burning is appropriate for the habitat (and where more specific conservation objectives do not apply), a general aim for heather management should be to encourage structural and species diversity between and within stands, and the presence of all growth phases of *Calluna*. This could provide significant biodiversity conservation benefits by providing habitats for fire-sensitive and slow colonising species, and species that are associated with mature and degenerate heather stands, which are currently largely absent in the uplands.

   However, this should not be achieved by simply lengthening the rotation period such that old heather is burnt, as this will risk poor regeneration, invasion by undesirable species and possibly long periods of bare soil and erosion. Instead a twin-track approach should be pursued. Firstly, most *Calluna* dominated stands should be carefully burnt in small patches and at a moderate frequency (when 20-30 cm high and in the building phase). In addition selected areas should be permanently set-aside from burning. Typically these should include all areas where burning is not recommended (eg along water courses, on steep ground and bogs) and stands of old heather (if present), but areas on level or sloping ground should also be included.

Where areas are taken out of burning management, livestock may need to be reduced or shepherded to avoid increased concentrations and grazing pressures on remaining areas under burning management.

8. Consideration should be given to permanently taking some large areas out of burning management, possibly at a landscape scale. Setting aside large areas from burning management (with appropriate fire break safeguards etc) may allow 'natural' succession processes to take place thereby increasing scrub and woodland cover in the uplands and habitat diversity in the long-term. The appropriateness, scale, priority and locations of such scrub and woodland expansion will depend on wider visions and conservation goals for the uplands. Such considerations will also need to take into account the desired spatial distribution and complimentarity of habitats, for example to allow for the colonisation and persistence of slowly dispersing species (both those that require disturbance and those that require undisturbed conditions).
9. Special efforts should be made to reduce the size and frequency of burns on *Molinia* and *Nardus* dominated grasslands. The total area burnt should also be limited to that which can be effectively grazed such that all or most of the seasons growth can be removed, thus avoiding rapid accumulation of litter and the need for further burning.
10. On heathlands and particularly bogs (if they must be burnt) burns should be carried out in conditions that limit burn temperatures to no more than that required to remove most woody vegetation, unless the creation of a bare seedbed is essential. 'Clean burns' are generally not necessary and should normally be avoided. On grasslands burns should be hot enough to remove the ground litter, but not hot enough to damage the regeneration abilities of any *Calluna* or other shrubs that are present.
11. Post-fire poaching and excessive grazing must be avoided, either by stock reductions / control or by ensuring sufficient areas are burnt to spread any grazing pressure.
12. Urgent consideration should be given to shortening the spring burning season to reduce the risks of severe and uncontrolled fires developing in dry conditions and potential nest losses of upland breeding birds. Consideration should also be given, after further research (see below), or trials, to extending the autumn burning season into September. Increasing the number of days over which burning may be carried out will reduce the pressure for burning in unsuitable conditions. Autumn burning also has several advantages over spring burning, but steps should be taken to ensure that autumn burning is not carried out in areas with high erosion risks. In general the autumn period produces fewer suitable burning days than the spring. Therefore, a longer extension of the autumn burning period would be required to compensate for any spring reduction.

7.4.2 Further research

From the information reviewed in this study it is evident that there are some key information gaps that constrain the evaluation of the impacts of burning on soils, hydrology and biodiversity. To address these gaps, it is suggested that further research should focus on the following issues in England (although many may also apply elsewhere in the UK).

1. Assessment of current burning practices in regularly burnt habitats in each of the main upland areas. This could be usefully combined with an assessment of the socio-economic

costs and benefits of burning management and the factors influencing current practices (such as available labour and expertise).

2. Confirmation and calibration of the applicability of previous burning research studies carried out on *Calluna* heaths in north-east Scotland in relation to *Calluna* dominated grouse moors in England. This is necessary because, for example, *Calluna* growth rates are likely to be much faster further south and west, whilst plant dispersal rates may not differ; and these factors may significantly influence post-fire vegetation re-establishment dynamics.
3. Detailed analysis of the overlap between the legal burning period and nesting periods of upland birds, and quantification of the overall risk of nest / egg loss through burning, taking into account the extent and frequency of burning in each species' specific nesting habitat.
4. Assessment of the advantages, disadvantages and overall impacts of extending the legal burning into September, with and without a shortening of the spring burning season.
5. Analysis of risks and potential impacts of wildfires in relation to any reduced management by burning (and/or grazing) of uplands.
6. Assessment of the erosion and hydrological impacts of management fires and wildfires eg by:
    - instrumentation of a small moorland catchment where there are regular management burns - measurement of flow, collection of suspended sediment by continuous-flow sampler(s) and separation into organic & mineral fractions. Intensive sampling programme to be synchronised with burning, less intensive sampling to capture seasonal patterns.
    - post-fire instrumentation of significant uncontrolled fire sites to collect flow and sediment data over a period spanning the early revegetation phases. Intensive measurements over the first significant rains to follow the fire.
7. Quantification of the impacts of burning on nutrient cycling and nutrient budgets, particularly on bogs and frequently burnt habitats (such as *Molinia* grasslands in south-west England).
8. Confirmation of the effects of fire on potentially fire-sensitive keystone species, such as *Sphagnum* species. This should assess fire survival abilities, but also establish if there are any species that are eliminated or severely reduced in the long-term as a result of the direct effects of fire or of environmental conditions immediately post-fire. Research should also attempt to distinguish these direct effects from the consequences that result from species merely having very slow dispersal rates.
9. Specific autecological studies of species of high conservation concern that may be at risk from current burning practices or changes in burning practices.
10. Assessment of the impacts of burning bogs on birds and invertebrate communities.
11. Research into the impacts of frequent burning of *Molinia* grasslands on biodiversity, particularly on the south-west moors.
12. Research into the impacts of accidental fires on upland calcareous grasslands.
13. Establishment of long-term large-scale exclosure experiments to examine the impacts of burning in the absence of grazing, and to assess vegetation changes in the absence of both.

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9. Species index

Note, this list excludes *Calluna vulgaris* due to its frequent mentions in the text, together with some other species listed in the text for which no information on burning impacts is presented.

10. Appendices

Appendix 1. List of consultees

English Nature staff

Dan Hunt - Kendal office (Cumbria)

Dave Clayden - York (N&E Yorks)

Dave Hazelhurst - Truro (Cornwall)

Eric Steer - Shrewsbury (N Mercia)

Flemming Ulf-Hansen - Taunton (Somerset & Glos)

Helen Stace - Ledbury (Hereford&Worcs)

John Barrett - Stocksfield (Northumbria)

Jon Hickling - Wigan (Lancs)

Jon Stewart - Bakewell (Peak District)

Paul Duncan - Wakefield (Pennines)

Peter Welsh - Leyburn (Yorkshire Dales)

Simon Bates - Exeter (Dartmoor)

National Park Administration Ecologists

Gill Thompson - Northumberland National Park

Judy Palmer - Yorkshire Dales National Park

Sue Rees – North York Moors National Park

Phil Taylor – Lake District National Park

Rhodri Thomas – Peak District National Park

Sue Goodfellow – Dartmoor National Park

Others

Des Thompson and Angus Macdonald - SNH

Barbara Jones - CCW Bangor

Paul Corbett - EHS Belfast

Dr Dave Baines - Game Conservancy

Dr Sarah Gardner – ADAS, Wolverhampton

Richard Brand-Hardy – DEFRA

Ian Condliffe - RDS Leeds

Dave Glaves – RDS Exeter

Steve Albon - CEH Banchory

Dr B. Emmett – CEH Bangor

Dr John Adamson – CEH Merlewood

Dr Rob Marrs – Liverpol University

Professor Robert Pakeman – Macaulay Institute

Dr David Gibbons – RSPB, Sandy

Dr Jeremy Wilson – RSPB, Edinburgh

Martin Gillibrand – Moorland Association

Simon Thorp (Heather (Trust)

John Holmes (Environment Agency)

Dr Colin Legg and Matt Davies (Edinburgh University)

Professor Tim Burt (Durham University)

Dr John Holden (Leeds University)

Alison Colls (University of East Anglia)

Professor Jules Pretty and Dr Peter Farage (University of Essex)

Appendix 2. Some research projects underway in 2003 that include an investigation of the impacts of fire on upland habitats in Britain

The following completed research questionnaires were received from the consultees listed in Appendix 1.

1. Study title: Carbon Sequestration in Upland Heather Moorlands
2. Lead organisation: Department of Biological Sciences / Centre for Environment and Society, University of Essex
3. Project manager at lead organisation: Professor Jules Pretty
4. Collaborating organisations: None
5. Funding bodies and programmes: N/A
6. Funding body project code (if any): N/A
7. Start date: May 2002
8. End date: October 2003
9. Overall aim of the study: To investigate the carbon sequestration potential of upland heather communities in the UK.
10. Specific objectives of the study: • Determine the quantity of organic matter lost from the system through burning • Estimate the total quantity of carbon stored in the vegetation and upper soil layers • Estimate net primary production • Quantify carbon efflux from the soil • Quantity the major nutrients within the system • Produce a carbon budget for the system and determine the effect of burning cycles on this budget
11. Outline of study methods: Within different heather age classes: • Above ground vegetation sampled using randomly assigned quadrats • Soil cores and blocks taken for determination of belowground biomass, soil nutrients and soil bulk density • Standard procedures used for determination of dry matter, carbon content, nutrient analysis • Soil carbon dioxide efflux measured using infrared gas analysis

1. Study title: The role of nature conservation in mitigating climate change
2. Lead organisation: Tyndall Centre for Climate Change Research
3. Project manager at lead organisation: Alison Colls (PhD researcher) supervised by Prof. Andrew Watkinson (UEA); Prof Bill Sutherland (UEA) and Mike Harley (English Nature). Melvin Cannell (CEH Edinburgh) is also involved in the project.
4. Collaborating organisations: (see funding consortium below)
5. Funding bodies and programmes: PhD co-funded by the Tyndall Centre and English Nature, representing Scottish Natural Heritage, Countryside Council for Wales, Environment and Heritage Service - Northern Ireland and The Woodland Trust.
6. Funding body project code (if any):
7. Start date: July 1 2002
8. End date: June 30 2005
9. Overall aim of the study: To examine the tradeoffs and synergies associated with incorporating carbon management into UK nature conservation policy and recommend win-win solutions.
10. Specific objectives of the study: Examine the greenhouse gas and biodiversity consequences of land use changes associated with habitat restoration and creation. Focus on several habitat and/or policy based case studies: • saltmarsh recreation and coastal realignment • deforestation of plantations to recreate open-ground semi-natural habitats • restoration of planted ancient woodland sites • impact of reduced grazing in the uplands
11. Outline of study methods: Interdisciplinary, integrated assessment of existing land use, soil, carbon and biodiversity databases. Anticipate making use of decision support tools such as cost benefit analysis and multi-criteria analysis. May use some carbon modelling techniques.

1. Study title: Fire behaviour and characteristics on heather moorland
2. Lead organisation: The University of Edinburgh
3. Project manager at lead organisation: PhD student: Matt Davies; Supervisor: Dr Colin Legg.
4. Collaborating organisations: Game Conservancy Trust (this is a NERC CASE studentship). Scottish Natural Heritage (additional financial contribution).
5. Funding bodies and programmes: NERC
6. Funding body project code (if any): NERC reference NER/S/C/2001/06470A (SNH code is F02AC204)

7. Start date: January 2002

8. End date: December 2004

9. Overall aim of the study:

The use of fire as a vegetation management tool is wide spread, yet little is known of the factors determining the behavioural characteristics of fire in dwarf-shrub vegetation. Understanding this form of moorland management has become increasingly important in the UK given the regional and national decline of heather cover and its contribution to the biodiversity of the uplands, and the importance attached to organic soils in carbon budgets and the regulation of climate. This project will determine the functional relationship between fuel, site and environmental characteristics and the ecological role of the fire through combustion of litter, damage to soil horizons and impact on regeneration of the vegetation.

10. Specific objectives of the study:

Empirical fire behaviour models have been developed, particularly in North America and Australia, but almost exclusively for forest or grassland vegetation and for the primary purposes of fire control. The relatively homogenous vegetation of heather moorland and seasonal burning practice provide the ideal situation for designed experimental fires to extend the theoretical basis of these models. These models will be adapted by this project for prediction of the ecological impact of fire on dwarf-shrub communities. The new understanding will greatly extend our knowledge of how fire can be manipulated to develop its ecological role and could substantially benefit conservation and economic management objectives.

In describing fuel and fire behaviour in quantitative terms, the project will address the hypotheses that the ecological impacts of fire are related to fire intensity and that the impacts can be predicted from a knowledge of the way fuel and weather characteristics affect fire behaviour.

11. Outline of study methods:

The project will, through novel research, provide a protocol for formal description of fuels in dwarf-shrub and grassland vegetation. This will include biomass estimation, a formal description of fuel structure and estimation of fuel moisture and combustibility of individual fuel components. Experimental fires will be used to relate fuel characteristics to fire-line intensity and heat production.

Experimental fires will be within an area including the Monadhliath and Grampian mountain ranges in Strathspey. The area will allow statistically valid site selection and replication for experimental fires and extensive informal replication within the normal population of management fires. Baseline habitat and historical grouse population data are also available for the area. Landscape scale analysis of mosaics will be on an individual moor scale using a GIS.

12. Location(s) of study (eg of experimental plots) and dominant habitat types present (define as UK BAP Broad Habitats types or NVC communities):

Monadhliath and Grampian mountain ranges in Strathspey.

BAP – Upland heathland

Natura – European dry heath; possibly Northern Atlantic wet heath with Erica tetralix.

13. Summary of progress / findings to date:

The project is still in its early stages, but methods have been developed for describing fuel characteristics. Experiments are underway to establish the relationship between fuel moisture and weather conditions. The first few experimental fires have been instrumented, though more information will be required before data can be analysed.

14. Reports / papers produced to date: None

15. Reports and papers in preparation or press: None

16. Form completed by:

Colin Legg, SGS, The University of Edinburgh.

Angus MacDonald, Scottish Natural Heritage

1. Study title: Demonstration Moors Project
2. Lead organisation: Heather Trust
3. Project manager at lead organisation: Simon Thorp, Director
4. Collaborating organisations: RDS DEFRA, English Nature Regional Offices, relevant National Parks, landowners, agricultural tenants, sporting tenants. A wider Steering Group is being established for each moor that will incorporate other bodies, groups and individuals in the area.
5. Funding bodies and programmes: DEFRA, with input in kind from landowners and occupiers
6. Funding body project code (if any): BD 1223
7. Start date: 1 May 2001
8. End date: 30 April 2006
9. Overall aim of the study: Establish a suite of 4 demonstration sites across England and Wales
10. Specific objectives of the study: • Enhance the sustainable dwarf shrub communities • Maintain and improve bio-diversity • Maintain and improve the productivity of domestic animals • Demonstrate on a farm scale the steps necessary to convert 'white' moorland dominated by matt-grass *Nardus stricta* and purple moor-grass *Molinia caerulea* to grass / heath mixtures • Improve the standards of pest control insofar as this affects the moorland vertebrates • Change the sheep and cattle husbandry system, where appropriate, in such a way that sheep ticks *Ixodes ricinus* (if present) are disadvantaged. • Hold open & training days on up to 20 days per year over all moors, but not exceeding 4 per moor per year. Every moor will host at least one day each year.
11. Outline of study methods: The purpose of the project is to demonstrate elements of best practice to a wide audience. The development of best practice on each moor will allow comparison of improvements on areas included within the project area outside the project. Scientific monitoring of each moor will take place to provide baseline data to compare with data collected during the course of the project.
12. Location(s) of study (eg of experimental plots) and dominant habitat types present (define as UK BAP Broad Habitats types or NVC communities): • Molland Moor, Exmoor. Heather and acid grassland. • Gwerclas Mountain, Corwen, North Wales. Heather. • Peak District site – to be confirmed. • Faulds Brow, Caldbeck Common

13. Summary of progress / findings to date: The project has been established on two moors and work to improve the condition of the moors is underway. The difficulties of work at Caldbeck Common have been well documented. Control of grazing will be the key element, and in particular further reduction of winter grazing levels. The project has had to withdraw from the first peak district site and negotiations are well advanced to find a replacement.
14. Reports / papers produced to date: None
15. Reports and papers in preparation or press: None
16. Form completed by: Simon Thorp

1. Study title: Mapping Countrysports Project
2. Lead organisation: The Game Conservancy Trust
3. Project manager at lead organisation: Dr Julie Ewald
4. Collaborating organisations: The National Gamekeeper's Organisation (including The Moorland Gamekeepers Association), The Scottish Gamekeepers Association, The Game Conservancy Trust, Countryside Alliance, The Council of Hunting Associations, The Anglers Conservation Association.
5. Funding bodies and programmes: Contributions from the above organisations, as well as from private individuals.
6. Funding body project code (if any):
7. Start date: Pilot in 1999; Main project started in 2001.
8. End date: End of 2003.
9. Overall aim of the study: Determine the amount of habitat management undertaken in support of countrysports throughout Great Britain and relate this to published information on the likely effect of this management on biodiversity in the wider countryside.
10. Specific objectives of the study: 1. Collate the area under the control of those who undertaken management for countrysports. 2. Determine the amount of habitat management undertaken on these areas in detail: for example area of heather moor burnt in a year, area of heather cut, type and area of game cover crops planted, coppicing of woodland undertaken for fox hunting, bankside fencing installed and maintained along river banks. 3. Relate this management to measures of biodiversity or habitat quality available from published sources: for example breeding bird distributions, records of rare wildlife etc.
11. Outline of study methods: • Locate countrysports managers and mail them questionnaires and maps for filling. • Enter returned data onto database/GIS. • Overlay mapped data with available info-such as mapped data available through CIS, EN etc. • Use published studies to relate returned information to effect of the management over the whole country.
12. Location(s) of study (eg **of experimental plots) and dominant habitat types present (define as UK BAP Broad Habitats types or NVC communities):** Great Britain

13. Summary of progress / findings to date: Shooting information – all mailings complete. Processing returns and filling in missing information.
14. Reports / papers produced to date: Some reporting in Game Conservancy Trust Annual Reviews
15. Reports and papers in preparation or press: Final Report at end of 2003, Two papers in preparation, including one relevant to moorland burning, on moorland management and distribution of breeding upland waders.
16. Form completed by: Dr Julie Ewald

1. Study title: Soil and soil properties as indicators of past and present vegetation conditions in upland environments
2. Lead organisation: University of Reading
3. Project manager at lead organisation: PhD student: Elizabeth Rees; Supervisors: Dr. Stephen Nortcliff (Dept of Soil Science) and Prof. Valerie Brown (Centre for Agri-Environmental Research).
4. Collaborating organisations: University of Reading Dept. of Soil Science Joint Nature Conservation Committee With support from English Nature and CEH, Merlewood
5. Funding bodies and programmes: University of Reading Joint Nature Conservation Committee
6. Funding body project code (if any):
7. Start date: October 2002
8. End date: October 2005
9. Overall aim of the study: To date within the United Kingdom consideration and evaluation of habitat conservation has focused almost exclusively on the nature of the above ground vegetation and associated macro- and meso-fauna. Whilst it is readily acknowledged that these are closely associated with the nature of soils there has been almost no consideration of the value of soils in determining nature conservation policies and priorities. This study seeks to investigate the nature of soil and soil properties under semi-natural upland moorland vegetation and the changes that occur in response to a range of short and long term land management practices on these moorlands.
10. Specific objectives of the study: This study will look at the properties of peat under unburnt, healthy vegetation and the properties of peat under vegetation burnt at differing intervals, both with and without grazing to determine if and how these properties differ. It will then look to answer a series of questions: • Do soil properties vary in their resilience to change? • Once the thresholds for change have been achieved, how quickly do soil property changes take place? • Are the changes reversible, and if so at what rate does the reversal take place? • At what stage do changes become irreversible?

11. Outline of study methods: The study will make use of a long-term experiment set up in 1954 to study the effects of burning and grazing management on the vegetation at Moor House. These include plots that have been burnt on 10-year rotation, 20-year rotation and not burnt since 1954. Previous soil survey work at Moor House has led to the classification of peat deposits based upon the phytosociological communities they support. This study will obtain comprehensive soil profile descriptions for the sites identified by phytosociological analysis. As such the soils will be described according to the procedures of the Soil Survey of England and Wales, and a range of soil biological, chemical and physical properties will be determined. Data analysis of results will allow the nature of the soils under differing management regimes to be determined. Information from the first phase of the project on the changes in soil properties and the rate of changes in these properties will provide the basis for the second phase, the development of models to predict the changes in soil properties in response to management. Upland moorland communities at other sites in the UK will provide the opportunity to test the predictions of the model.
12. Location(s) of study (eg of experimental plots) and dominant habitat types present (define as UK BAP Broad Habitats types or NVC communities): Moor House – Upper Teesdale NNR, North Pennines Upland heath and blanket bog
13. Summary of progress / findings to date: Due to commence field work May 2003
14. Reports / papers produced to date: None
15. Reports and papers in preparation or press: None
16. Form completed by: Elizabeth Rees

Appendix 3. Glossary

Fire terms are taken from *Prescribed burning on Moorland: Supplement to the Muirburn Code: A Guide to Best Practice* (Anonymous, 2001).

Aerial fuels	Standing or supported plant material not in contact with the ground.
Alpine	Above the limit of tree growth (see also Montane).
AONB	Area of Outstanding Natural Beauty
Available fuel	The amount of fuel which will actually be consumed under particular weather conditions and fire behaviour. Usually much less than either potential fuel or total fuel.
Back-fire, or backing fire	Fire spreading against the wind direction.
Broad habitat	A framework classification developed under the UK BAP process for habitat types across the whole of the UK
Burning rotation	The period of time between one fire and the next within a particular burning unit.
Burning unit	A specific patch to be burnt.
Canopy	The foliage and fine twigs of dwarf-shrubs (such as heather), shrubs or trees.
Common Standards Monitoring	A framework of standards agreed by the UK Conservation agencies and JNCC for the monitoring of statutory sites in the UK (ie SSSIs, SPAs and SACs)
Creeping fire	Slowly spreading fire with short flames
CROW Act	Countryside and Rights of Way Act 2000
cSAC	Candidate Special Area of Conservation
DEFRA	Department for the Environment, Food & Rural Affairs
EA	Environment Agency
EN	English Nature
ESA	Environmentally Sensitive Area
EU	European Union
Favourable Condition	According to Common Standards Monitoring, the target condition for an interest feature in terms of the abundance, distribution and/or quality of that feature within the site, that the aim is to attain.
Fine fuels, or flash fuels	Fine material no more than a few millimetres in thickness (6 mm or less is frequently used internationally), which dries quickly and is usually sufficiently dry to burn within a day of rain (and possibly within as little as a couple of hours for some types of fine fuel).
Fire front	The line or strip along which there is continuous flames.
Fireline intensity	The rate of heat release per unit length of fire front per unit of time. Also known as Byram's fire intensity.

Fuel moisture content	Water content of potential fuel as a percentage of oven dry weight.
Ground fire	Fire burning in the soil, usually in peat or duff
Ground surface intensity	The total amount of heat energy released per unity area at the ground surface. This is probably of greater ecological significance than fireline intensity (see above).
HAP	Habitat Action Plan
Head (of the) fire	The front edge of a fire where the intensity and rate of spread are greatest.
Head-fire, or heading fire	A fire which spreads in the same direction as the wind.
Humus	Amorphous material formed by the partial decomposition of plant and animal remains; representing the organic constituent of soil.
Inbye	Enclosed grassland that is not normally ploughed or reseeded within a period of ten years, is used for pasture, hay or silage production and is subject to regular inputs of fertiliser.
Interest Feature	According to Common Standards Monitoring, a habitat, habitat matrix, geomorphological or geological exposure, a species or species community or assemblage which is the reason for notification of the site [SSSI, SAC or SPA] under the appropriate selection guidelines or, in the case of Natura 2000 and Ramsar areas, the features for which the site will be designated.
LFA	Less Favoured Area
LU	Livestock Unit: dairy cow = 1.0 LU; beef cow (excluding calf) = 1.0 LU; cattle over 2 years old = 1.0 LU; cattle from 6 months to 2 years = 0.6 LU; Lowland ewe and lamb = 0.15 LU; hill lamb = 0.10 LU; Ram and tegs over 6 months = 0.15 LU; horses and ponies = 1.0 LU. Source: MAFF 2001b.
MAFF	Ministry of Agriculture, Fisheries and Food
Montane	Above the potential tree-line, which is c. 600 m in England and synonymous with alpine (but on the continent the term includes the sub-alpine zone).
Muirburn	The term used for the management of moorland vegetation by burning in Scotland
Needle ice	A form of ground ice that consists of groups ice slivers at or immediately below the ground surface. Needle ice is about a few centimetres long.
NNR	National Nature Reserve
NVC	National Vegetation Classification
Plant litter	The cast leaves, shoots, and fragments of bark, flowers, seed capsules and other dead or dying plant material which falls to the ground.
Potential fuel	The material which might burn in the most intense fire likely on a site. Usually less than total fuel.
Prescribed fire	A planned and controlled fire set under specified conditions to achieve specified land management objectives.
Priority Habitat	Habitats identified as being a priority in the UK BAP (see www.ukbap.org.uk)
Priority Species	Species identified as being a priority in the UK BAP (see www.ukbap.org.uk) because they are globally threatened and/or rapidly declining in the UK

RDS	Rural Development Service (was FRCA until April 2001)
Regolith	Loose earth material above the underlying solid rock.
Residence time	The time during which flaming combustion is occurring above a fixed point at the surface of the fuel (or ground). The time taken for the fire front to pass.
SAC	Special Area of Conservation, designated under the EU Habitats Directive (EU Council Directive 92/43/EEC on the conservation of natural habitats of wild fauna and flora).
SAP	Species Action Plan
Semi-improved	"Grassland which has been modified by the application of fertilisers (generally at low level over a long period of time), herbicides, intensive grazing or drainage such that its species-richness and diversity is lower that that of unimproved semi-natural grassland but still retains some characteristics of the semi-natural grassland from which it has been derived" (Backshall *et al* 2001).
Semi-natural	Plant associations often created by direct or indirect effects of man.
Smouldering	Burning without flames and spreading very slowly.
SPA	Special Protection Area, designated under the EU Wild Birds Directive (*Council Directive 79/409/EEC on the conservation of wild birds*).
Species of Conservation Concern	Species listed as being of conservation concern in the UK BAP.
SSSI	Site of Special Scientific Interest
Surface fire	Fires in grass, heather, shrubs, plant litter.
Swaling	The term used for the management of vegetation by fire in the south-west of England.
Total fuel	The total live and dead plant material above the mineral soil, including any peat or humus layer above the mineral soil. May include material which will rarely, if ever, burn except under the most extreme circumstances.
UKBAP	The UK Biodiversity Action Plan. Although no single document exists that is called the UK BAP, this terms is widely used to refer to the total programme of actions identified as part of implementation of the UK Biodiversity Strategy (see www.ukbap.org.uk).
Unfavourable Condition – recovering	According to Common Standards Monitoring, an interest feature can be recorded under the condition category recovering after damage if it has begun to show, or is continuing to show, a trend towards Favourable Condition. This category can be recorded more than once for a particular feature in relation to a single damaging activity.
Unimproved grassland	"Grassland that has not been subjected to agricultural improvement through the use of fertilisers, herbicides, intensive grazing or drainage. Such grasslands are often species-rich." (Backshall *et al*, 2001)
Wildfire	An uncontrolled fire burning outside the limits of prescription, or a fire which has escaped control.

Appendix 4. National Vegetation Classification (NVC) communities (Rodwell 1991a, b, 1992, 1995, 2000) referred to in this report

Broadleaved, mixed and yew woodland / coniferous woodland

W19 *Juniperus communis ssp. communis – Oxalis acetosella*

W21 *Crataegus monogyna – Hedera helix* scrub

Calcareous grassland

CG2 *Festuca ovina – Avenula pratensis* grassland

CG9 *Seslaria albicans – Galium sterneri* grassland

CG10 *Festuca ovina – Thymus praecox* grassland

CG11 *Festuca ovina – Agrostis capillaris – Alchemilla alpina* grass heath

Acid grassland

U4 *Festuca ovina – Agrostis capillaris – Galium saxatile* grassland

U5 *Nardus stricta – Galium saxatile* grassland

U7 *Nardus stricta – Carex bigelowii* grass heath

U10 *Carex bigelowii – Racomitrium lanuginosum* moss heath

U17 *Luzula sylvatica – Geum rivale* tall herb community

U21 *Crytogramma crispa – Deschampsia flexuosa* community

Dwarf shrub heath

H4 *Ulex gallii – Agrostis curtisii* heath

H9 *Calluna vulgaris – Deschampsia flexuosa* heath

H10 *Calluna vulgaris – Erica cinerea* heath

H12 *Calluna vulgaris – Vaccinium myrtillus* heath

H13 *Calluna vulgaris – Cladonia arbuscula* heath

H16 *Calluna vulgaris – Arctostaphylos uva-ursi* heath

H18 *Vaccinium myrtillus – Deschampsia flexuosa* heath

H19 *Vaccinium myrtillus – Cladonia arbuscula* heath

H21 *Calluna vulgaris – Vaccinium myrtillus – Sphagnum capillifolium* heath

Mires

M1 *Sphagnum auriculatum* bog pool community

M3 *Eriophorum angustifolium* bog pool community

M5 *Carex rostrata – Sphagnum squarrosum* mire

M8 *Carex rostrata – Sphagnum warnstorfii* mire

M9 *Carex rostrata – Calliergon cuspidatum/giganteum* mire

M10 *Carex dioica – Pinguicula vulgaris* mire

M13 *Schoenus nigricans – Juncus subnodulosus* mire

M15 *Trichophorum cespitosum (Scirpus cespitosus) – Erica tetralix* wet heath

M11 *Carex demissa – Saxifraga aizoides* mire

M16 *Erica tetralix – Sphagnum compactum* mire

M18 *Erica tetralix – Sphagnum papillosum* mire

M19 *Calluna vulgaris – Eriophorum vaginatum* blanket mire

M20 *Eriophorum vaginatum* blanket and raised mire

M25 *Molinia caerulea – Potentilla erecta* mire

M26 *Molinia caerulea – Crepis paludosa* mire

M37 *Cratoneuron commutatum – Festuca rubra* spring

M38 *Cratoneuron commutatum – Carex nigra* spring.

Inland rock

OV37 *Festuca ovina – Minuartia verna* community

OV38 *Gymnocarpium robertianum – Arrhenatherum elatius* community

OV39 *Asplenium trichomanes – Asplenium ruta-muraria* community

OV40 *Asplenium viride – Cystopteris fragilis* community

Appendix 5. Scientific names of animals referred to in this report

Amphibians

Common Frog *Rana temporaria*

Palmate Newt *Triturus helveticus*

Reptiles

Vivaparous lizard Lacerta vivapara

Birds

Taken from *The British List* as published by British Ornithologists Union on their website (accessed on 3rd April 2003).

Wigeon *Anas Penelope*

Teal *Anas crecca*

Red Grouse *Lagopus lagopus*

Black Grouse *Tetrao tetrix*

Hen Harrier *Circus cyaneus*

Buzzard *Buteo buteo*

Golden Eagle *Aquila chrysaetos*

Kestrel *Falco tinnunculus*

Merlin *Falco columbarius*

Peregrine Falcon *Falco peregrinus*

Dotterel *Charadrius morinellus*

Golden Plover *Pluvialis apricaria*

Lapwing *Vanellus vanellus*

Dunlin *Calidris alpine*

Snipe *Gallinago gallinago*

Curlew *Numenius arquata*

Redshank *Tringa totanus*

Red-necked Phalarope *Phalaropus lobatus*

Short-eared Owl *Asio flammeus*

Sky Lark *Alauda arvensis*

Meadow Pipit *Anthus pratensis*

Dipper *Cinclus cinclus*

Whinchat *Saxicola rubetra*

Stonechat *Saxicola torquata*

Northern Wheatear *Oenanthe oenanthe*

Ring Ouzel *Turdus torquatus*

Carrion Crow *Corvus corone*

Hooded Crow *Corvus cornix*

Common Raven *Corvus corax*

Twite *Carduelis flavirostris*

Mammals

Rabbit *Oryctolagus cuniculus*

Mountain hare *Lepus timidus*

Short-tailed Vole *Microtus agrestis*

Red Fox *Vulpes vulpes*

Pine Marten *Martes martes*

Stoat *Mustela erminea*

Weasel *Mustela nivalis*

Polecat *Mustela putorius*

Badger *Meles meles*

Wildcat *Felis silvestris*

Appendix 6. Terrestrial UK BAP Priority Habitats found within the uplands of England that are subject to burning, their current resource and conservation / restoration targets

Targets were taken from Habitat Action Plans (HAPs) on the UK BAP website (www.ukbap.org.uk) on 10 January 2003.

Priority Habitat	Current resource	Key UK BAP targets
Blanket bog	215,000 ha of blanket peat soil in England	Maintain the current extent and overall distribution of blanket mire currently in favourable condition. Improve the condition of those areas of blanket mire which are degraded but readily restored, so that the total area in, or approaching, favourable condition by 2005 is 340,000 ha (ie around 30% of the total extent of restorable blanket mire). Introduce management regimes to improve to, and subsequently maintain in, favourable condition a further 280,000 ha of degraded blanket mire by 2010. Introduce management regimes to improve the condition of a further 225,000 ha of degraded blanket mire by 2015, resulting in a total of 845,000 ha (ie around 75% of the total extent of restorable blanket mire) in, or approaching, favourable condition.
Limestone pavement	3,000 ha in the UK, largest areas occurring in North Yorkshire and Cumbria	Ensure that there is no further loss to the extent of limestone pavement areas. Revised Ensure that there is no further deterioration in the quality of existing limestone pavement areas. Revised Maintain features of geological importance Revised Restore and maintain a characteristic assemblage of native plant species Revised
Upland calcareous grassland	10,000 ha	Maintain the current distribution and extent Achieve favourable condition for at least 75% by 2005 Initiate pilot attempts to re-create at least 200 ha of upland calcareous grassland by 2005, with a particular emphasis on reducing fragmentation.
Upland heathland	270,000 ha	Maintain the current extent and overall distribution of the upland heathland, which is currently in favourable condition. Achieve favourable condition on all upland heathland SSSIs/ASSIs by 2010, and achieve demonstrable improvements in the condition of at least 50% of semi-natural upland heath outside SSSI/ASSIs by 2010 Increase dwarf shrubs to at least 25% cover where they have been reduced or eliminated. A target for such restoration of between 50,000 and 100,000 ha by 2010 is proposed. Initiate management to re-create 5,000 ha of upland heath by 2005 where heathland has been lost due to agricultural improvement or afforestation, with a particular emphasis on reducing fragmentation of existing heathland.

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Top left: Using a home-made moth trap.
Peter Wakely/English Nature 17,396
Middle left: Co_2 experiment at Roudsea Wood and Mosses NNR, Lancashire.
Peter Wakely/English Nature 21,792
Bottom left: Radio tracking a hare on Pawlett Hams, Somerset.
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