Controlled burns for Red Grouse, at Howden Moors, Peak District © David Claves/Natural England

The effects of managed burning on upland peatland biodiversity, carbon and water (NEER004)

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Natural England Evidence Review NEER004

The effects of managed burning on upland peatland biodiversity, carbon and water

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External expert review group

A review group comprising three external national experts, chaired by a senior Natural England scientist, oversaw the topic review. The group was also attended by David Glaves (Senior Upland Specialist, Natural England). The group was responsible for ensuring that the topic review included all relevant and appropriate evidence, that the sifting and assessment of papers had been carried out in line with the methodology and that the conclusions correctly interpreted the evidence. The review group have confirmed that this was the case. The review group members were:

- Dr Mike Morecroft (chair), Natural England
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Cover photograph

Controlled burns for Red Grouse, at Howden Moors, Peak District © David Glaves/Natural England

Executive summary

Management of the English uplands is complex and achieving good environmental outcomes, while taking into account the needs of owners, stakeholders and other interests is a balancing act. An uplands evidence review has been undertaken in which a number of candidate topics have been considered. These topics were identified through stakeholder input, reflection on areas of advice subject to challenge and looking at what could make a difference on the ground. The five priority topics identified have formed the review programme and will help further the understanding of available evidence to support uplands management.

This topic review focused on a series of questions which were evaluated against scientific evidence. The topic review has also helped identify areas for future research; in the next phase, beyond the review programme, additional relevant information will be considered, for example social and economic factors, current working practices and geographic scale. The evidential conclusions drawn from these additional areas will help inform our future advice and practical management of the uplands on the ground.

Context

Peatlands are areas with a naturally accumulated layer of carbon-rich peat, formed from dead and decaying plant material under waterlogged conditions. The United Kingdom (UK) is of international importance for peatlands, holding between 9-15% of Europe's peatland area and about 13% of the world's blanket bog. Upland peatlands in England comprise two UK Biodiversity Action Plan (BAP) priority habitats: blanket bog and upland flushes, fens and swamps, together with the wet heath element of upland heathland, that generally occur on unenclosed land above the Defra Moorland Line.

Burning is widely used as a tool in the management of a range of moorland vegetation types including upland peatlands, principally:

- to create new growth for livestock grazing;
- to increase the diversity of the age and structural of heather[1] for game management; and more recently
- for conservation management.

In England, upland peatland habitats are often degraded, with the characteristic, varied mire vegetation 'modified', in some cases to the point that it is dominated by a single species, particularly heather or purple moor-grass and, in places, hare's-tail cottongrass (Critchley 2011a, Defra 2011). This reflects a variety of past impacts including those of atmospheric pollution, overgrazing, drainage and burning, including wildfires (UK Biodiversity Group 1999, Natural England 2010, Defra 2011). More recently, concerns have been expressed by some about the possible effects of burning on aspects of biodiversity associated ecosystem services, especially carbon sequestration, and water quality and flow, although the effects are much debated (Yallop *et al.* 2009, IUCN 2011, Worrall *et al.* 2011, Holden *et al.* 2012).

Purpose

The purpose of this topic review is to review the available evidence on the effects of managed burning on upland peatland biodiversity, carbon sequestration and water quality and flow.

[1] Scientific names of species mentioned in the Executive summary are given at first mention (and subsequent mentions for plants) in the main text of the report.

Scope

The topic review covers biodiversity maintenance and restoration objectives, including the effects of burning on modified, degraded upland peatland habitats and their restoration, and the effects on carbon sequestration, and water quality and flow, key ecosystem services. Biodiversity is the variety of life and has many different aspects, but in the present context we use it to refer to the habitats and characteristic associated species of flora and fauna of upland peatland habitats, in particular blanket bog, and associated upland habitats on peat soils, including flushes, fens and swamps, and wet heath. Characteristic flora and fauna are those species associated with less modified, functioning, high quality upland peatland habitats.

Restorability per se is not covered (although it is considered in the UER blanket bog restoration topic report, Shepherd *et al.* 2013).

This topic review sets out the evidence base; it does not make recommendations about how this evidence should be interpreted and applied to Natural England's working practice and advice. Consideration of other relevant information, such as practicality of implementation (on which social and economic considerations have a bearing), landscape and archaeology/historic environment, is an important part of the process of developing advice, but is not part of this uplands evidence review programme.

The search for evidence was confined to temperate and boreal peatlands (especially blanket bog, but including other bog/mire/fen/wet heath), biodiversity (flora and fauna), carbon sequestration, water (quality and flow), and (managed) burning. References relating to dry heath, mineral soils, forests, tropical/arctic/tundra areas and wildfire (unless related to the effect of management burning) were excluded with a few exceptions when they provided relevant evidence that was otherwise missing.

Questions addressed by the evidence review

The over-arching topic review question is:

What are the effects of managed burning on the maintenance and restoration of upland peatland biodiversity, carbon and water?

The following sub-questions provide a further focus for the review:

a) What are the effects of managed burning on the maintenance and restoration of the characteristic floristic composition, structure and function of upland peatland habitats?
b) What are the effects of managed burning on the maintenance and enhancement of the characteristic fauna of upland peatlands either directly or indirectly through changes in vegetation composition and structure?
c) What are the effects of managed burning of upland peatlands on carbon sequestration, either directly or indirectly through changes in vegetation composition and structure?
d) What are the effects of managed burning of upland peatlands on water quality (including colouration, release of metals and other pollutants and aquatic biodiversity) and water flow (including downstream flood risk), either directly or indirectly through changes in vegetation composition and structure?
e) How do differences in the severity, frequency, scale, location and other characteristics of burns (including 'cool burns') affect upland peatland biodiversity, carbon and water?
f) How does the interaction of managed burning and grazing affect upland peatland biodiversity, carbon and water?
g) Is there a relationship between managed burning of upland peatlands and 'wildfire' (risk, hazard, occurrence, severity and extent)?
h) What is the extent, frequency, practice and type of managed burning (including 'cool burning') on upland peatlands (including in relation to designated sites and water catchments)?

Due to the multiple factors being considered under most of the sub-questions, evidence relating to discrete elements within these sub-questions was assessed separately. Many references contributed evidence to multiple sub-questions.

Process

An initial literature search and a call for evidence from stakeholders produced a list of 895 references (excluding duplicates). Filtering on title and abstract reduced this list to 492 references that were likely to be relevant and these were obtained and assessed against inclusion-exclusion criteria. As a result of this process, 227 papers were accepted for quality assessment and summarising, with 170 references grouped in to 123 evaluated studies used in the topic review.

Summary of conclusions

The nature and strength of the evidence relating to the effects of managed burning was reviewed for each sub-question and from this, evidence statements and the following conclusions were developed. A total of 54 evidence statements were developed from the evidence derived from evaluated studies. Of these, the majority were classed as strong, 19 (35%), or moderate, 28 (52%), with only six (11%) classed as weak and one (2%) as inconsistent. There were clear differences in the volume and strength of evidence across the eight sub-questions with a greater volume and stronger evidence on the effects on flora, fauna, carbon and water, and on burning extent, but less on the effects of differences in severity, interactions with grazing and wildfire. Recommendations for future research and other evidence gathering to address gaps are made at the end of this summary.

a) What are the effects of managed burning on the maintenance and restoration of the characteristic floristic composition, structure and function of upland peatland habitats?

There is strong evidence that managed, rotational burning results in a change in the species composition of blanket bog and upland wet heath vegetation, at least for a period of time. This included strong evidence that:

- Burning of blanket bog and wet heath typically leads to an initial period of graminoid dominance, in particular of hare's-tail cottongrass, purple moor-grass or deergrass, typically lasting 10-20 years, and with an initial decline in dwarf-shrub cover and in some cases diversity.
- Heather and some other dwarf shrubs tend to decline during the initial graminoid-dominant phase, but typically then increase, especially on drier sites, and may become dominant. This may take 15-20 years or longer on less-modified, wetter blanket bog and may not occur, for example, with too frequent or severe burning and/or heavy grazing.
- Bryophytes as a group tend to decline initially after burning of blanket bog. *Sphagnum* bog-mosses as a group have shown mixed responses, in some cases increasing in the early post-burn stages, sometimes declining or being killed and sometimes then increasing or recolonising after varying periods.
- Burning is associated with the creation of bare ground at least at a fine-scale.

There is moderate evidence that:

- Burning leads to an increase in cloudberry.
- The composition of blanket bog vegetation can continue to show change more than 80 years after the last burn.
- Burns can lead to the creation of relatively flat, unpatterned bog surfaces. This may be followed by the re-establishment of hummock-hollow topography following gradual recovery or recolonisation of *Sphagnum* bog-mosses.

There is relatively little evidence on the effects of differences in burning rotations on peatlands, with only one long-term experimental study (at Moor House National Nature Reserve in the North

Pennines) that has covered multiple rotations of differing lengths (10 and 20 years). This provides moderate evidence that differences in frequency of burning affect the vegetation composition and structure of blanket bog. At this site, more frequent burning has promoted dominance of hare's-tail cottongrass, with heather achieving higher cover under the longer rotation.

Changes in vegetation composition and structure may affect the functioning of the peatland ecosystem and hence have effects on associated ecosystem services which are reviewed in subsequent sub-questions. When interpreted in relation to the characteristic floristic composition, structure and function of upland peatland habitats, overall these vegetation responses to burning, in particular the tendency to dominance of graminoids and/or heather may reduce the chance of maintaining active, functioning peatland. Similarly, where restoration is an objective for modified, degraded upland peatland habitats, burning may perpetuate dominance of graminoids or heather.

b) What are the effects of managed burning on the maintenance and enhancement of the characteristic fauna of upland peatlands either directly or indirectly through changes in vegetation composition and structure?

There is strong evidence that burning indirectly influences the invertebrate community composition of upland peatland habitats, typically benefiting open-ground species such as ground beetles and surface-active spiders. Many of the studies indicate an increase in overall species-richness or diversity and suggest that this occurs through increases in structural diversity at a relatively fine scale and the presence of open patches or short swards amongst taller unburnt vegetation resulting from patchwork burning. Thus, conditions are provided for open ground species and for species that favour taller vegetation such as some web-spinning spiders. Many of the studies were carried out on modified upland peatlands and hence not all of the invertebrate species and assemblages are necessarily characteristic peatland species associated with less modified, functioning, and high quality upland peatland habitats.

There is moderate evidence, that too frequent burning is likely to render peatland sites less suitable or unsuitable for the large heath butterfly (a UK BAP species), but that occasional burning may be beneficial perhaps in favouring the larval foodplant, hare's-tail cottongrass, and in reversing succession on at least some drier sites.

There is moderate evidence (reviewed under sub-question d) that burning is correlated with changes in the diversity and composition of aquatic invertebrate assemblages in watercourses draining upland peatland catchments. These changes reflect declines in certain groups, especially mayflies and stoneflies, and increases in flies.

There is strong evidence of correlations between moorland habitat types, their vegetation composition and structure, and densities of some moorland breeding birds, particularly waders. In few studies has this been related directly to peatlands rather than moorland in general or specifically to burning practice. We can however say that there is strong evidence that:

- Certain species are associated with particular moorland vegetation characteristics. Red grouse and stonechat are associated with increasing heather cover; snipe and curlew with heterogeneity in vegetation structure; golden plover and skylark with short vegetation; waders with wet conditions; whinchat with dense vegetation; stonechat with tall vegetation; and meadow pipit with grass-heather mixes.
- There are correlations between burning and/or predator control intensity and densities of some moorland breeding birds. Higher densities of red grouse, golden plover and curlew with increased burning/predator control were each shown in two studies; and higher densities of lapwing, redshank and ring ouzel each in single studies. Two studies showed lower densities of meadow pipit and single studies showed lower densities of skylark, wheatear and twite with increasing intensity of burning/predator control.
- The dates of first egg-laying of some moorland bird species and the legal burning season (which closes on 15 April in the uplands in England) overlap in the first half of April. In all but one case, it was the minority of first nest attempts (with nine species having more than

10% of first egg-laying attempts before 15 April). This indicates a potential vulnerability for those species that nest on the ground or in vegetation that is likely to be burnt (six species) rather than actual losses. There is also moderate evidence of earlier nesting over time for eight species, which may in the future increase the proportion of first nest attempts by mid-April.

Only one study looked in detail at changes in numbers of breeding birds (for five waders) in relation to burning. This showed moderate evidence of greater declines in golden plover under more intensive (rather than less intensive) burning management and that curlew and lapwing declined more on 'heather-dominated' plots than on 'bog' plots.

One study showed moderate evidence of an increase in breeding success and numbers of lapwing, golden plover, curlew and red grouse and breeding success of meadow pipit in response to legal predator control, indicating that such control contributes to the increases shown by some species on grouse moors in other studies probably in addition to any burning effects.

There is weak evidence of a correlation between burning and/or predator control intensity and overall diversity of moorland breeding birds.

c) What are the effects of managed burning of upland peatlands on carbon sequestration, either directly or indirectly through changes in vegetation composition and structure?

There is strong evidence that managed burning affects various components of the carbon cycle of upland peatlands. This includes strong evidence that:

- Moorland burning results in increased water colouration and/or dissolved organic carbon (DOC) in peatland watercourses.

There is moderate evidence that:

- Burning reduces peat accumulation and reduces above and below ground carbon storage compared to no burning.
- Managed burning can result in erosion and reduction in the level of the soil surface.
- There are increases in gross CO_2 fluxes of respiration and photosynthesis.
- There are carbon loses through fuel consumption during burning and in conversion to char.

Only relatively recently have attempts been made to estimate complete carbon budgets that consider the overall impacts of burning. So far, these have produced inconsistent evidence, with predictions of both positive and negative overall effects of burning, although the estimates provide strong evidence that burning affects the processes controlling carbon budgets of upland peatlands.

d) What are the effects of managed burning of upland peatlands on water quality (including colouration, release of metals and other pollutants and aquatic biodiversity) and water flow (including downstream flood risk), either directly or indirectly through changes in vegetation composition and structure?

As noted above (c), there is strong evidence that moorland burning results in increased water colouration and/or dissolved organic carbon (DOC) in peatland watercourses. Related to this there is:

- Strong evidence that the area of recent burning on deep peat is correlated with an increase in water colouration and/or DOC at the catchment-scale in watercourses draining peatland catchments.
- Moderate evidence that the area of heather-dominated vegetation on deep peat is correlated with an increase in water colouration and/or DOC, in soil water in one case and in watercourses draining peatland catchments in another.

- Moderate evidence from laboratory studies that burning is associated with an increase in water colouration and increased pH (which are likely to be related as pH controls solubility of DOC).

However, the relatively small number of recent small plot- or stand-scale studies of water colouration and/or DOC in relation to burning have shown inconsistent evidence. It has been suggested that this may reflect differences in time since burning (as effects have been shown to be greatest soon after burning) and sampling too deep in the peat (as effects have been shown to occur only in the upper layer).

In relation to soil and water chemistry, there is weak evidence of differences in concentrations of chemical entities after a burn, for example, with aluminium, iron and sodium increasing and calcium, chlorine, bromine and pH declining.

There is weak evidence from small-scale plot/stand studies: of shallower water tables initially after burning; and of increased frequency of surface runoff after recent burning. However, no evidence was identified specifically relating to the effect of burning on watercourse flow or the risk of downstream flood events. If there are any effects, they are likely to be highly site specific.

As noted under sub-question (b), there is moderate evidence that burning is correlated with changes in the diversity and composition of aquatic invertebrate assemblages in watercourses draining upland peatland catchments. These changes reflect declines in certain groups, especially mayflies and stoneflies, and increases in flies.

e) How do differences in the severity, frequency, scale, location and other characteristics of burns (including 'cool burns') affect upland peatland biodiversity, carbon and water?

Few studies were identified that related differences in the severity and other characteristics of burns directly to differences in effects on the four aspects of upland peatlands reviewed under the main sub-questions (a-d). However, there is:

- Strong evidence that moisture content, vegetation type and phenology, recent weather and human factors are important factors in the ignition of fires.
- Moderate evidence that fuel load and structure are critical factors in fire behaviour, particularly in 'fireline' intensity (heat output per unit length of fire front) and rate of spread, although residence time and depth of penetration of lethal temperatures into the soil are perhaps more important in determining severity of impact, but are much less well understood.

Little evidence was identified on the types of burning practice taking place in the English uplands in general and specifically on deep peat, including on the extent to which 'cool burning' is practised. However, there is moderate, recent evidence (reviewed under sub-question h) that burns into the bryophyte and lichen layer occur in a proportion of cases on blanket bog and wet heath.

There is evidence that, in addition to initial fire severity, pre-fire vegetation composition is an important factor influencing post-fire recovery. Related to this, there is moderate evidence (reviewed under sub-question a) that differences in the frequency of burning (ie rotation or return period) affect vegetation composition and structure on blanket bog. More frequent burning tends to promote dominance of single competitive species, particularly graminoids, especially hare's-trail cottongrass and purple moor-grass, and in drier situations, heather.

Some of the catchment-scale studies (reviewed under sub-question d) are relevant to burn frequency as they suggest that water colouration and/or DOC are related to the area of recent burning which is determined by rotation length; the shorter the rotation, the greater the proportion that has recently been burnt. Thus, there is strong evidence that increased frequency of burning results in an increase in water colouration and/or DOC.

No evidence was found that specifically identified differences in effects related to differences in the size or location of burns on upland peatlands, although more generally there is some evidence that larger fires tend to be more variable in terms of intensity and severity. They also reduce diversity at a fine scale in terms of vegetation composition and structure compared to a mosaic of different-aged burns, although this relates particularly to heather and heaths.

f) How does the interaction of managed burning and grazing affect upland peatland biodiversity, carbon and water?

From the relatively small number of evaluated studies that included grazing treatments, there were few significant interactions between burning and grazing, although there are many studies that demonstrate significant effects of these two major moorland management practices separately (see also the UER topic report on the impacts of moorland grazing and stocking rates, Martin *et al.* 2013). It is however possible that interactions may occur at a relatively large, for example, moorland grazing unit, scale and are not easy to pick up in smaller plots. For example, new growth, particularly of graminoids, following burning generally attracts stock. Thus, burning is specifically used for stock management to provide more even grazing. The extent, including the size and distribution of burn patches, as well as total area burnt, can influence the distribution and level of grazing by stock. However, there is:

- Moderate evidence from the only long-term burning and grazing experiment (at Moor House) of some interactions with grazing in the initial period following burning: in particular a greater increase in the extent of bare ground and an increase in grazing on, and reduction in cover of, cloudberry, compared to ungrazed treatments.
- Moderate evidence that burning results in increased grazing of purple moor-grass by sheep and deer, but that this may be short-lived.
- Weak evidence that burning on short rotations and/or heavy grazing after burning can lead to maintenance of the dense graminoid phase in wet heath (rather than its replacement by heather), but that high grazing intensity and low burning frequency pushes the balance in favour of heath rush and mat-grass.

g) Is there a relationship between managed burning of upland peatlands and 'wildfire' (risk, hazard, severity, extent and damage etc)?

Although little evidence was found of a direct relationship between managed burning and the occurrence, severity and extent of wildfire, there is moderate evidence that fuel load and structure are critical factors in fire behaviour. Burning reduces fuel load and may therefore have benefits for fire risk management, alongside other measures such as cutting and/or the creation of a network of firebreaks and control zones. There may be an increased need for fire risk management in future, if certain climate change scenarios become a reality.

There is moderate evidence that 'heather moorland' in the Peak District, which is mostly managed by rotational burning, is less prone to the occurrence of wildfires than other moorland habitats. Although they still occur relatively frequently. This reflects the large extent of heather moorland and lower than predicted occurrence in relation to area.

h) What are the extent, frequency, practice and type of managed burning (including 'cool burning') on upland peatlands (including in relation to designated sites and water catchments)?

There is strong evidence from a recent national mapping exercise that rotational management burning occurs on about a quarter of the total moorland deep peat resource in England, mostly in the Pennines, Bowland and Northumberland. Over the remainder of the resource, burning on deep peat is now infrequent or does not normally take place, although it did occur on some of this land in the past.

There is moderate evidence that there has been an increase in the extent (and frequency) of managed burning on moorland in England, including specifically on degraded ‘dry’ blanket bog in the North Peak ESA. There is also moderate evidence of a recent increase in the number of gamekeeper’s employed and potential number of shooting days per year (both 29% between 2001 & 2009) on grouse moors in the north of England, although this relates to all heather-dominated moorland rather than specifically peatlands.

There is moderate evidence that there is considerable variation in the frequency of burning on upland peatlands:

- Nationally, the average burn ‘return period’ (for heather-dominated proportion in brackets) was 64 year (26.5 year). However, this includes the majority of upland peatland that is subject to little or no rotational burning.
- In areas where rotational burning occurs on deep peat, the proportion burnt per year and hence the average return period is higher than the national average: 21.2 year (11.7 year) in the North York Moors, 39.3 year (15 year) in the North Pennines and 73.1 year (25 year) in the Peak District (and 25% in another earlier study of heather-dominated ‘dry bog’ in the North Peak ESA).
- In most of these areas the proportion burnt per year on upland peatland and dry heath are similar. And
- The proportion burnt per year in the national dataset was similar on Sites of Special Scientific Interest (SSSI) and on (the smaller area of) non-designated upland peatland.

No evidence was identified on the coincidence of burning and water catchments.

There is little evidence on the types of burning practice taking place in the English uplands in general and specifically on deep peat, including on the extent to which ‘cool burning’ is practiced. There is, however, moderate evidence that burns into the bryophyte and lichen layer and in to sensitive areas occur in a proportion of cases on blanket bog and wet heath (in 11-17% of all, including unburned, samples in two national surveys).

Research recommendations

Assessment of the available evidence indicates that the following areas would benefit from further research:

- The extension of experimental and other monitoring studies of the effects of burning on vegetation and ecosystem services to a wider range of sites across the English upland peatland resource, ideally including additional medium/long-term studies covering multiple rotations across the full length of typical blanket bog burn rotations (for example, 15-25 year) (which are currently restricted to the Hard Hill experiment at Moor House). Ideally these should consider type of burning (for example, ‘cool’ and other burns). Such studies should also include the wider range of upland peatland habitats including wet heath, flushes, fens (including valley mires) and swamps, and consider the interaction of burning and grazing across the range of typical stocking rates and regimes that occur in moorland grazing units that include peatland habitats.
- Research on post-burn recovery times in upland peatlands including palaeo-archival studies on vegetation recovery after fire. Research on the effects of burning on the range of characteristic upland peatland species, especially individual *Sphagnum* bog-moss species, including post-burn recovery.
- Improved, more detailed and consistent description of the characteristics of study sites, for example, in terms of habitat, degree of modification, vegetation composition (including *Sphagnum* species) and structure, surface topography and condition, not just in vegetation but in wider studies, for example, on carbon and water. In addition, also recording information about the type and ideally intensity and/or severity of burns in related research projects.

- Improved and more consistent interpretation of existing and new vegetation data from an ecological and nature conservation/biodiversity perspective, for example, including consideration of aspects of autecology, functional types and associations, disturbance, habitats and vegetation community types, habitat condition, associated species, structure (including micro topography) and function.
- Research on restoration management, including the potential use of one-off burning and alternative treatments to reduce graminoid and heather dominance where this is an objective.
- Research on the effects of burning on key characteristic blanket bog species of fauna particularly invertebrates, reptiles and birds (including food availability, for example, craneflies as an important food item for waders).
- Further examination of data on bird nesting dates and breeding success in relation to burning (for example, from Nest Record Cards, vulnerability/risk from burning (especially short-eared owl and stonechat) and pre-nesting activity timing).
- Further studies addressing the relative lack of information on gaseous exchange of peatlands in relation to burning and on char production during burning and its significance.
- Extension of studies on aquatic invertebrates more widely across the English uplands. Interpretation of changes in community composition in terms of water quality and biodiversity, possibly including as food availability for predators (for example, fish and birds such as dipper).
- Studies of the effects of differences in the intensity/severity of fires and characteristics of burn patches such size, shape, location (for example, in relation to slope, watercourses etc), distribution etc
- National collation of data on the occurrence and characteristics of wildfires, including the relationship with managed burning and further study of the occurrence of wildfire in relation to managed burning on upland peatlands, perhaps by extending the modelling work done in the Peak District.
- Repeat of remote sensing surveys to map changes in the extent and frequency of burning on upland peatlands, particularly blanket bog, nationally and in the main areas where burning occurs in the north of England.
- Definitive, agreed mapping of grouse moors, together with data on burning management, for correlation studies, particularly with breeding bird survey data, and the relationship to other land uses including water catchments and designated sites.
- Improved recording of the occurrence and severity/effects of burning and wildfires in site surveys of upland peatland habitats, for example in Natural England's condition assessment/'integrated site assessment'. National collation and analysis of data from Natural England's condition/integrated monitoring surveys particularly in relation to burning-related attributes. A repeat of the national sample survey of more detailed condition assessment of upland habitats in the Priority Habitat Inventories (last completed in 2008-10), perhaps on a rolling programme with a proportion of new sites added to the existing sites.

Contents

Appendices

List of tables

1 Introduction

Background

1.1 In March 2011 Defra published the Government's review of uplands policy which sets out a range of actions the Government, led by Defra, will take in partnership with others in the public, private and voluntary sectors to help secure a sustainable future for the English uplands. The actions in the Uplands Policy Review sit under four main themes:

- Supporting England's hill farmers.
- Delivering public goods from upland environments (including biodiversity).
- Supporting sustainable upland communities.
- Driving and monitoring change.

1.2 Natural England has a specific role in helping deliver the Uplands Policy Review, in particular through our research and evidence based advice, our delivery of agri-environment schemes and our partnership work with the hill farming and moorland management sector and rural communities to deliver a wide range of public goods and environmental benefits. Our role in the uplands is also shaped by our broader role in the delivery of the Government's Natural Environment White Paper and Biodiversity 2020 aspirations that focus on the enhancement and protection of ecosystem services and the natural environment, including improving the condition of England's Sites of Special Scientific Interest (SSSI). Biodiversity 2020 targets for SSSIs are to achieve 50% in favourable condition and maintain 95% in favourable or recovering condition by 2020.

1.3 For these reasons it is important that our advice and decisions are based on sound evidence, and that our evidence processes are transparent and robust.

The need for the review programme

1.4 The English uplands are extensive and include a range of biotypes, species, and land management practices. It is widely recognised that they provide provisioning, regulatory, and cultural ecosystem services.

1.5 As such, the uplands present a number of environmental conservation and land management challenges. This is particularly the case in understanding the effects of land management operations on upland biodiversity and ecosystem services.

1.6 The Uplands Evidence Review (UER) programme draws together the best available science to provide sound evidence on the effects of land management activities on upland biodiversity and ecosystem services. In doing so it provides a basis for advice and decisions on future management of the uplands.

The nature of the evidence

1.7 Over several decades a body of evidence has accumulated exploring the effects of different types of land management interventions on a range of upland ecosystem services, habitats and species. There is a wide variety of study types, for example 'before and after studies', correlation studies and survey/monitoring, which may have taken advantage of opportunities for natural experiments. Randomised control trials, which represent perhaps the most robust experimental approach are uncommon in the literature on burning and vary in the degree of replication and whether they are fully or partially randomised. There are many methodological differences within this literature, notably the lack of consistency between measurement

methods and different outcome measures. There are also limitations in the degree to which small plots with small-scale experimental burns represent the real world situation of the patchwork of rotational burns across large moorland sites. Neither can controlled treatment and observational studies take into account the range of factors that affect natural systems. Nevertheless the results provide a basis from which conclusions about the effects of interventions can be drawn and research needs can be identified.

1.8 It is worth noting a number of significant challenges associated with undertaking a review of the evidence concerning upland management interventions. Firstly, the search strategy needs to be broad enough to capture studies from a wide range of sources including journals not indexed in environmental databases, and work that may be in the 'grey' literature (such as reports or case studies). At the same time it is important to make a rigorous assessment of the quality of all sources and the extent to which they can be applied to the question under study. Finally, the wide range of study types, for example post-treatment only measures or uncontrolled pre- and post- studies, presents problems in making comparisons.

Overall scope of the Upland Evidence Review Programme

1.9 The uplands are a broad biotype encompassing a variety of habitat, species and ecosystem services that are subject to a variety of land management interventions. This review of biodiversity and ecosystem upland evidence will focus on five issues where there is significant challenge:

- The impact of tracks on the integrity and hydrological function of blanket peat.
- Restoration of degraded blanket bog.
- The effects of managed burning on upland peatland biodiversity, carbon and water.
- Upland Hay Meadows: what management regimes maintain the diversity of upland meadow flora and populations of breeding birds?
- Impact of moorland grazing and stocking rates.

1.10 This report presents the findings from the review of the effects of burning on upland peatlands.

1.11 Consideration of other relevant information, such as social and economic factors, landscape and archaeology/historic environment, is an important part of the process of developing our advice, but is not part of the uplands evidence review programme. The review does not consider the effect of the state of knowledge of the evidence on Natural England's position and advice.

1.12 The consideration of ecosystem services in this burning topic review is restricted to carbon and water, with other provisioning and supporting services not included in its scope. Whilst consideration of the likely future climate change scenarios is specifically excluded from the reviews programme, evidence on the interaction between managed burning and weather variables and upland peatland biodiversity and carbon and water ecosystem services has been considered.

Review topic: The effects of burning on upland biodiversity and ecosystem services

The issue

1.13 Burning is widely used as a tool in the management of a range of moorland vegetation types including upland peatlands, principally to create new growth for livestock grazing, and age

and structural diversity of heather *Calluna vulgaris*[2] for game (red grouse *Lagopus Lagopus scoticus*) management and, more recently, for conservation management . As noted in the heather and grass burning code (Defra 2007b),"when used with skill and understanding, (burning) can benefit agriculture, game birds and wildlife, but if it is used irresponsibly it creates risks for people, property and the environment."

1.14 Blanket bog is the most extensive upland peatland habitat with the UK holding an internationally important resource. But in England much of it, and other upland peatland habitats, is degraded, with the characteristic, varied mire vegetation 'modified', in some cases to the point that it has been replaced by dominance of single species, particularly *Calluna* or purple moor-grass *Molinia caerulea*[3] and, in places, hare's-tail cottongrass *Eriophorum vaginatum*. (Critchley 2011a, Defra 2011). This reflects a variety of past impacts including atmospheric pollution, overgrazing, drainage and burning, including wildfires (UK Biodiversity Group 1999, Natural England 2010, Defra 2011). More recently, concerns have been expressed by some about the possible effects of burning on biodiversity associated ecosystem services, especially carbon sequestration, and water quality and flow, although the effects are much debated (Crowle 2007, Yallop *et al.* 2009, IUCN 2011, Worrall *et al.* 2011, Holden *et al.* 2012). These issues are the subject of this topic review and are considered further in Section 3.

What is considered in this topic review?

1.15 The topic review covers biodiversity maintenance and restoration objectives, including the effects of burning on modified, degraded upland peatland habitats, and the effects on carbon sequestration and water quality and flow ecosystem services. Restorability per se is not covered but is considered in the UER restoration of degraded blanket bog topic (Shepherd *et al.* 2013).

The over-arching topic review question

1.16 **What are the effects of managed burning on the maintenance and restoration of upland peatland biodiversity, carbon and water?**

The following sub-questions were the focus of the topic review:

a) What are the effects of managed burning on the maintenance and restoration of the characteristic floristic composition, structure and function of upland peatland habitats?
b) What are the effects of managed burning on the maintenance and enhancement of the characteristic fauna of upland peatlands either directly or indirectly through changes in vegetation composition and structure?
c) What are the effects of managed burning of upland peatlands on carbon sequestration, either directly or indirectly through changes in vegetation composition and structure?
d) What are the effects of managed burning of upland peatlands on water quality (including colouration, release of metals and other pollutants and aquatic biodiversity) and water flow (including downstream flood risk), either directly or indirectly through changes in vegetation composition and structure?
e) How do differences in the severity, frequency, scale, location and other characteristics of burns (including 'cool burns') affect upland peatland biodiversity, carbon and water?
f) How does the interaction of managed burning and grazing affect upland peatland biodiversity, carbon and water?
g) Is there a relationship between managed burning of upland peatlands and 'wildfire' (risk, hazard, occurrence, severity, extent and damage)?

[2] Hereafter referred to as *Calluna*.
[3] Hereafter referred to as *Molinia*.

h) What are the extent, frequency, practice and type of managed burning (including ‘cool burning’) on upland peatlands (including in relation to designated sites and water catchments)?

1.17 The text for sub-questions (e) and (g) differs slightly from that given in the project scoping document (Natural England 2012) with ‘intensity’ and ‘impact’ replaced by ‘severity’ after adoption of the definitions proposed by Keeley (2009) in this topic review. These are given in the Glossary, but briefly, ‘severity’ relates to above- and below-ground organic matter consumption and impact. Fire ‘intensity’ relates to the energy output from fire so is still covered in this sub-question under other characteristics of burns. Other ‘fire behaviour’ terms are defined in the Glossary. In addition, in sub-questions (e) and (f) ‘ecosystem services’ has been replaced with ‘carbon and water’ to reflect the focus of the review on these two aspects of ecosystem services.

1.18 Definitions of key terms included in the above overarching question and sub-questions used in the review were:

- Peatland biodiversity comprises the habitats, habitat structure and characteristic associated species of flora and fauna of upland peatland habitats, in particular blanket bog, and associated upland habitats on peat soils, including flushes, fens and swamps, and wet heath. Characteristic flora and fauna are those associated with less modified, functioning, high quality upland peatland habitats.
- Ecosystem services included in the scope of this review are the carbon sequestration and storage aspects of climate regulation, and water provision (including water quality and especially colouration, release of metals and other pollutants and downstream aquatic biodiversity, especially invertebrates) and regulation (water flows and flood prevention). It should be noted that these are only part of a wider range of supporting, provisioning, regulating and cultural services that are beyond the scope of this review.

1.19 Further technical terms used in the report are defined in the Glossary.

1.20 Both English and scientific names of species are given at first mention in the text and subsequently English names are used (when available) for animals and scientific names for plants.

2 Methods

2.1 This chapter briefly sets out how this topic review was undertaken following the approach described *Natural England Evidence Reviews: guidance on the development process and methods* (Stone, 2013). There were some differences in detail from the guidance, in particular, the screening and assessments of references was generally done by one person rather than two. Quality assurance was carried out by a second person independently assessing a small proportion of references and reviewing the completed assessment checklist and evidence table forms. In addition, a sample of five references across the range of types and subjects was independently assessed by the three external experts in the topic review group and compared with the Natural England assessments. Overall, this revealed a good level of consistency across the assessment approach.

General principles

2.2 The review process systematically identified all available studies providing evidence for the specific questions posed. A long preliminary list of documents is sifted, to ensure that those that are included meet defined criteria ('inclusion criteria') and do not meet others ('exclusion criteria').

2.3 The 'PICO' framework provides a structured approach to formulating review questions and framing the over-arching search strategy (Stone 2013) so inclusion criteria can be objectively set. It derives from medical reviews and the terminology reflects this. It comprises the following four elements:

- Population: the population/species/habitat of interest, in this instance, upland peatland habitats in England.
- Intervention: the intervention, activity or approach to be used, in this instance, managed burning.
- Comparison: the main alternative to the intervention, in this instance, no burning, at least in recent decades; and/or a comparator, which in this instance was upland peatland biodiversity and ecosystem services prior to burning intervention or where burning has not occurred as far as is known in recent times (the past century or more).
- Outcome: the outcomes or effects that are being considered, in this instance, maintenance and restoration of biodiversity and delivery of ecosystem services.

2.4 The topic review provides a narrative overview of the evidence from included studies, with evidence statements providing a synthesis for each sub-question.

Evidence search

2.5 Literature searches were conducted using the terms listed below. References were downloaded, or manually added if necessary, into a reference manager database (EndNote Web) and duplicates removed. References were also identified through contact with key national experts and lead organisations, and from scrutinising relevant, recent reviews: Worrall *et al.* (2011), IUCN (2011), Bain *et al.* (2011), Holden *et al.* (2011, 2012), Grant *et al.* (2012) and Heinemeyer & Vallack (2012). In addition, there was an open call to interested stakeholders and others to submit evidence material for consideration as part of the review.

Search terms

2.6 Potential search terms were identified for each of the PICO categories in search plans for each of the eight sub-topic questions. The following search terms were used (an asterisk denotes a wild card search term allowing for several permutations of the term):

Blanket bog, Blanket peat, Blanket mire, Peat*, Bog*, Mire*, Upland*, Moorland*, Moor*, Peat depth, Bog pool*, Hummock*, Hollow*, Acrotelm, Catotelm, Peat* form*, Wet heath*, Fen*, Flush*, Swamp*, Raised bog*, Wildfire*, Fire*, Burn*, Managed burn*, Controlled burn*, Prescribed burn*, Cool burn*, Rotation*, Intensity, Frequency, Rotation, Location, Scale, Area, Size, Distribut*, Temperature, Risk*, Hazard*, Intens*, Extent, Damag*, Prevent*, Resilien*, Fuel load, Fuel-assisted, Water, Flow, Quality, Colouration, DOC, Pollut*, Metal*, Flood*, Infiltrat*, POC, Hazen*, Surface flow, Overland flow, Carbon sequestration, Carbon storage, Carbon balance, Carbon*, Biodiversity, Ecosystem*, Flor*, Fauna, Animal*, Bird*, Insect*, Invertebrate*, Habitat*, Communit*, Species, Composition, Frequency, Abundance, Diversity, Structure, NVC, Cover, Richness, *Sphagnum* moss*, Sphagn*, *Erica tetralix*, Eriophor*, *Molinia*, *Molinia* caerulea, *Calluna*, *Trichophorum*, Population*, Breeding success, Restor*, Revegetat*, Vegetation management, Remediation, Regenerat*, Livestock, Stocking, Grazing, Designated site*, SSSI, Site* Of Special Scientific Interest, Water catchment*, Special Area* Of Conservation, SAC*, Special Protection Area*, SPA, Non-designated, Catchment*, Mapping, Aerial, Satellite, Remote sensing, Land cover, Habitat mapping, GPS, Survey*.

2.7 These were used in ‘search strings’ that normally included one or more terms from each or several of the PICO categories, normally including the population and intervention (ie burning or related terms).

Search strategy

2.8 The following databases were searched:

Web of Science from 1990, CAB Abstracts from 1990, Zoological Record from 1978, Google Scholar and Scirus.

2.9 Publication searches were undertaken on:

2.10 British Library EThOS, Collaboration for Environmental Evidence (CEE), CCW library catalogue, COPAC, Global peatland restoration manual, Natural England library catalogue, water@leeds, PeatNet, Peatscapes and United Utilities/SCaMP.

2.11 The open call for evidence attracted 14 submissions from stakeholders and individuals: the CLA, Exmoor the Federation of Yorkshire Commoners, Moors for the Future, National Park Authority, the GWCT, the National Sheep Association, the National Trust, RSPB, United Utilities, the University of Leeds, Yorkshire Water and the following individuals: Roy Brown, A. E. Peart and Adrian R Yallop. The submissions were compared against references already selected for assessment and any additional papers and reports from the submissions were included in the assessment process. Other, non-evaluated, information submitted will be considered in a subsequent Guidance Refresh project.

Selection of references for inclusion

2.12 The number of references included in each part of the study is given in Table 1. The search strategy resulted in 895 titles (excluding duplicates). These were screened by title and abstract for relevance. In total 492 were assessed to be relevant and the full papers were retrieved and checked against the inclusion/exclusion criteria. In general references that did not specifically relate to burning were excluded. However, to address the potential indirect effects of burning on vegetation composition and structure in relation to sub-topic questions (b) (fauna), (c) (carbon) and (d) (water), references relating to the effects of changes in

vegetation composition and structure were included for these sub-questions. References on the autecology of plant species were not included other than as context. A large number of reviews and guidance documents were identified. Only the major review documents that were considered directly relevant to the sub-topic questions were formally assessed. Many of the remaining reviews and the guidance documents were, however, included as context in Section 3 and the introduction and context sections to the individual sub-question sections of this report. Where any uncertainty existed the full paper was assessed by a second reviewer. A total of 227 references were accepted for quality assessment, 170 of which were used in the report which were grouped for presentation and analysis as 123 studies.

2.13 The inclusion criteria used in screening references were: temperate and boreal peatland (especially blanket bog, but including other bog/mire/fen/wet heath), biodiversity (flora and fauna), carbon, water (quality and flow), and (managed) burning. The exclusion criteria used in screening references were: dry heath, mineral soils, forest/woodland/trees, tropical/arctic/tundra and wildfire (unless related to the effect of management burning). In a few cases, references that met the exclusion criteria were included, often in consultation with the review group, where they provided information relevant to the topic questions that would otherwise not have been included. The topic review group reviewed lists of included references at several stages and make suggestions for the inclusion of missing references and, in a few cases, for the exclusion of included references.

Table 1 Number of references included in each stage of the review

Review stage	Number of references
References captured using search terms in all sources (including duplicates)	1,980
References captured using search terms in all sources (excluding duplicates)	895
References remaining after title filter	645
References remaining after abstract filter	492
References remaining after full text filter	227
References used in topic review report	170[1]

[1]Grouped into 123 individual 'studies' for evaluation.

Study type and quality appraisal

2.14 Each study was categorised by study type (categorised as type 1-4, Table 2) and graded for quality against criteria appropriate for the study types (area/population, method of allocation to intervention or comparison, outcomes and analyses for quantitative studies and theoretical approach, study design, data collection, trustworthiness, analyses and ethics for qualitative studies) using a code: ['++'], ['+'] or ['-'] (Table 3), based on the extent to which potential sources of bias had been minimised.

Table 2 Types of studies

Rating	Definition
1	Meta-analyses, systematic reviews of, or individual Randomised Control Trials (RCT).
2	Systematic reviews of, or individual, non-randomised controlled trials, case-control trials, cohort studies, controlled before-and-after (CBA) studies, interrupted time series (ITS) studies, correlation studies, site comparisons and surveys.
3	Non-analytical studies, for example, case reports and case series studies.
4	Expert opinion and formal consensus.

2.15 Studies were quality appraised against quality criteria appropriate for the study types and subsequently classified into one of three categories (Table 3) in relation to the overarching burning topic- and sub-questions rather than necessarily the reference as a whole.

Table 3 Quality categories of studies

Rating	Definition
++	All or most of the methodological criteria have been fulfilled. Where they have not been fulfilled the conclusions are thought very unlikely to alter (low risk of bias).
+	Some of the criteria have been fulfilled. Those criteria that have not been fulfilled or not adequately described are thought unlikely to alter the conclusions (risk of bias).
-	Few or no criteria have been fulfilled. The conclusions of the study are thought likely or very likely to alter (high risk of bias).

Study categorisation

2.16 This section presents a summary of the type, quality, location and duration of the 123 studies included in the topic review. Similar information is given for each individual study in Appendix 1.

2.17 Fourteen studies (11%) were categorised as type 1, 80 (65%) as type 2, four as type 3 (3%) and 26 (21%) as type 4 (Table 4). In terms of quality, 24 (19%) were classed as [++], 69 (56%) as [+] and 31 (25%) as [-]. The main reasons for studies being assessed as [-] quality were: poor descriptions of sites, especially vegetation and soils, and whether upland peatland habitats were included or separated from other moorland habitats and hence unclear applicability to the review topic; and poor or unclear representativeness. Other reasons included: failure to describe methods adequately; low quality measure(s) of ecosystem and biodiversity outcomes; lack of clarity that bias had been minimised; and failure to take potential confounding factors into account.

2.18 Some of the studies evaluated included multiple interventions or other elements and/or addressed more than one sub-question. In a few cases this meant that different elements of a study differed in their quality and when this was the case, different quality scores were assigned to the relevant sub-questions. A complete listing of evaluated studies is available in Appendix 1.

Table 4 Categorisation of type and quality of studies included in the review

Study type	Study quality			Total
	++	+	-	
1	3	10	1	14
2	21	37	22	80
3	0	3	1	4
4	0	18	7	25
Total	24	68	31	123

Note: where studies were given more than one type/quality category (relating to different aspects of the study), the highest class is given in the table.

Description of studies

2.19 The 123 studies and their findings are described in Sections 4-11 of the report and in more detail in Appendices 3-11. They comprise the following study sub-types (based on a single main type for each study):

- 50 site comparisons/chronosequence studies;
- 24 reviews;
- 18 survey/monitotoring/aerial photographic interpretation studies;
- 12 randomised/partially randomised control trials;
- 9 models/carbon budget studies;
- 4 laboratory studies;
- 3 case studies; and
- 3 systematic reviews/critical syntheses.

Country of studies

2.20 The majority of evaluated studies (excluding laboratory studies and those where origin was unknown) were undertaken in the UK, involving 109 primary studies, with no more than two from other countries, all but one in Europe (Table 5). Of the UK studies, the majority were from England (77) with fewer from Scotland (17), 2 from Wales, 1 from Northern Ireland, and the remainder crossing country boundaries and including English sites or samples. Within England, only one of the studies included sites away from the northern uplands, in the SW uplands. In the north, most studies were undertaken in, or included sites in, the Pennines (44), with far fewer including sites elsewhere: Northumberland (outside N Pennines, 5 studies) and North York Moors (4). Within the Pennines, most studies included sites in the North Pennines and Peak District (both 21 studies), with fewer in the Yorkshire Dales (8) and South Pennines (4). More than half of the North Pennine studies (12) included sites at Moor House National Nature Reserve (NNR). All but one of the 24 reviews (including systematic reviews and critical syntheses) were undertaken in the UK and mainly involved UK data (Table 5).

Table 5 Number of evaluated studies by country of origin

Country of origin	Number of primary studies	Number of reviews
UK	107	23
England	75	-
Scotland	17	-
Wales	2	2
N Ireland	1	-
Ireland	2	-
France	1	-
Germany	1	-
Finland	1	-
Belarus	1	-
USA	1	1

Note: that there is some overlap between countries as some studies cover more than one country, and between primary studies and reviews, as some studies included both.

Duration of outcome measures

2.21 Where study length was given (excluding reviews), six studies measured short-term outcomes up to 12 months (post-burn follow up in studies that directly related to burning interventions, though this was not the objective of all evaluated studies). The majority of studies (42) measured outcomes over 1-2 years, with 11 covering 3-5 years, nine 6-10 years and only four greater than ten years. These follow up periods are short compared to typical burning rotations of 15-25 years on upland peatlands, although as noted above, some studies were not directly related to post-burn outcomes. The only long-term burning study was the Hard Hill burning and grazing experiment at Moor House National Nature Reserve which was established in 1954. The British Trust for Ornithology's Nest Record Scheme also includes long-term data on the timing of egg-laying dating back to 1939.

Assessing applicability

2.22 Each study was assessed on its external validity: that is, whether or not it was directly applicable to the target habitats and setting(s) in the scope, ie upland peatlands in the UK and particularly England. This assessment took into account whether the study was conducted in the UK, how representative it was of the English upland peatland resource as a whole and any barriers identified by studies or the review team. In order to make this assessment requires a comparison with a general assessment of the characteristics and state of upland peatlands in England. The reviews by Natural England (2010) and JNCC (2011) were used for this purpose.

Synthesis

2.23 It was not appropriate to use meta-analysis to synthesise the outcome data as interventions, methods and outcomes were heterogeneous. This topic review is restricted to a narrative overview of all studies that met the inclusion criteria and contained sufficient data for quality assessment. The studies were examined within the effects identified by the sub-questions. The evidence statements were developed using:

- The best available evidence of the effect of an intervention including the study type. With a few exceptions where reviews contributed (which are specifically noted in the text), this was from primary studies.
- The quality and quantity of supporting evidence and its applicability to the areas/populations and settings in question.
- The consistency and direction of the evidence.

2.24 On the basis of these factors, the strength of the evidence on particular aspects where there was clear evidence of an effect was classed as: strong, moderate, weak or inconsistent. This is partly a subjective judgment taking into account the above factors, though the following definitions were used as guidance:

- **Strong**: a large number of studies (typically >4-5) showing consistent trends or one or two high quality or national, representative studies [1++, 1+ or 2++].
- **Moderate**: a smaller number (at least two-three) of studies of which at least one was classed as a minimum of [2+].
- **Weak**: one study or a low number of generally lower quality studies, usually including some or most classed as minus [-].

3 Context on upland peatlands and burning practices

3.1 This section provides a general description of upland peatland habitats and burning practices to provide context to the issues described under the individual sub-questions. This (and in the 'introduction and context' sections in Sections 4-11 for individual sub-questions) and the references quoted are intended to aid interpretation and consideration of the findings of the review of evidence presented in the 'studies and evidence' parts of subsequent Sections 4-11. They do not form part of the evidence review itself or feed into evidence statements and conclusions. The section also describes the approach followed in presenting the evidence in subsequent sections.

Upland peatland habitats and ecosystem services

3.2 Uplands have been defined in various ways, including by: altitude (>250-300 m AOD); plant and animal habitats, communities or assemblages, for example by Countryside Survey 2000 land cover classes; geographically or topographically, such as by National Character Areas; and by use or disadvantage, such as by the Less Favoured Area (LFA), in particular the Severely Disadvantaged Area (SDA). Moorland is usually defined as semi-natural unenclosed land in the uplands which generally supports wet and dry dwarf-shrub heaths, blanket bog and other mires, scrub, bracken and grassland and, more locally, montane and rocky habitats including limestone pavement. The extent of moorland in England was mapped for MAFF (now Defra) in 1992 (ADAS 1993). This area, often referred to as within or above the Moorland Line, covers 7,990 km^2 (36% of the LFA).

Upland peatland habitats

3.3 Peatlands are areas with a naturally accumulated layer of carbon-rich peat, formed from dead and decaying plant material under waterlogged conditions (Bain *et al.* 2011). Mosses, mainly *Sphagnum* bog-moss species are the main peat formers in the UK. Natural peatlands are characterized by having a two-layered structure: the living surface layer, the acrotelm, composed mostly of recently deposited material with a matrix of growing plants especially *Sphagnum;* and the base layer, the catotelm, which remains permanently waterlogged and anaerobic. Carbon is sequestered and peat is formed in the acrotelm, whereas in the catotelm decomposition is very slow and the layer acts as a passive surface layer (Bain *et al.* 2011).

3.4 The UK is of international importance for peatlands, holding between 9-15% of Europe's peatland area (46,000-77,000 km^2) and about 13% of the world's blanket bog (Bain *et al.* 2011). There are various estimates of the extent of peatland soils and habitats in England, with the latest for upland peatlands being 3,553 km^2 of upland deep peat derived from blanket bog and upland valley mire (Natural England 2010). This is considerably larger than the area included in the priority habitat inventory for blanket bog: 2,553 km^2 (Natural England 2008) reflecting differing information sources. No figures are available for the extent of flushes, fens and swamps and wet heath in England, although they are smaller than the extent of blanket bog. Sixty-nine percent (1,761 km^2) of the area of blanket bog in the priority habitat inventory is designated as SSSI and the majority of this also as Special Areas of Conservation (SAC) (Natural England 2008).

3.5 Upland peatlands comprise two UK BAP priority habitats: blanket bog and upland flushes, fens and swamps, together with the wet heath element of upland heathland (UK Biodiversity Group 1999, Maddock 2008). They generally occur on unenclosed land above the Moorland Line. Blanket bog is a type of mire habitat 'fed' solely by atmospheric precipitation (from which it also receives nutrients) and found on blanket peat of between 0.3 and typically 6 m deep

(Lindsay 2010). In 'active' peat-forming condition, blanket bog vegetation is dominated by peat-forming species such as bog-mosses, cottongrasses and other sedges. There can be considerable variation in the surface patterning and micro-topography of blanket bogs, with hummocks, ridges, *Sphagnum* lawns, wet hollows and pools. Dwarf shrubs may be present on the drier areas, for example, the hummocks. Modified, degraded forms tend to lack such surface microtopography and become dominated by graminoids, particularly *Eriophrum vaginatum*, *Molinia* and Deergrass *Trichophorum cespitosum* or in drier situations, *Calluna*. Thus, the vegetation may resemble 'dry heath-type' vegetation but on blanket peat substrate rather than mineral soil. There is no agreed threshold for the minimum depth of peat that can support blanket bog vegetation communities and topography. The range of vegetation community types that comprise upland peatland habitats are given in paragraphs 4.2-4.11.

3.6 In 2008, 70% of blanket bog in England was regarded as in favourable or recovering condition (Natural England 2008). Since that time, the percentage in recovering condition has increased largely as a result of increased uptake of Higher Level Stewardship (HLS) agreements on moorland, so that currently 88% is in recovering condition and 11% in favourable condition (Natural England unpublished data).

Ecosystem services

3.7 There has been increasing interest in the ecosystem services provided by the natural environment and there is a growing understanding of the value of the benefits to society that arise from a high quality environment, none more so than from peatlands, most notably blanket bog. These cover both the provisioning and regulating services. Amongst those of particular importance in relation to upland peatlands are: carbon sequestration and storage in relation to climate regulation; drinking water supply and the quality of that water (with a significant proportion (70%) of the UK's drinking water coming from the uplands); and the potential for flood mitigation and alleviation. These are the subject of this review and are described further in Sections 6 and 7.

Burning management on upland peatlands

3.8 Fire is widely regarded as a natural component of heathland ecosystems (for example, Tucker 2003, MacDonald 1999, Rackham 1986 and Webb 1986) although natural fires resulting from lightening strikes are a relatively rare occurrence (Davies *et al.* 2008). Fire has been used by man since at least the Mesolithic period and is considered by some to have played a part in the initiation of the development of blanket mire in some areas (Moore 1973, 1975, Charman 2002). The modern English landscape, especially in the uplands, has been extensively exposed to management burns as well as uncontrolled 'wildfires' (natural and accidental). On blanket bog this has probably particularly been the case from the eighteenth century with the intensification of sheep production followed by game management for red grouse from the nineteenth century (Simmons 2003). Increasing livestock numbers in the uplands in response to government support in Less Favoured Areas in the latter part of the last century probably also resulted in an increase in burning of upland peatland and other moorland habitats at that time (Lindsay 2010).

3.9 Burning is widely used as a tool in the management of a wide range of vegetation types principally to create new growth for livestock grazing, and age and structural diversity for game (red grouse) management. More recently, it has been used in habitat management for conservation purposes particularly to create and maintain structural diversity, especially in dry heaths (Backshall *et al.* 2001, Defra 2007). It is also used following outbreaks of heather beetle *Lochmaea suturalis* with the aim of restoring damaged or killed *Calluna* stands (Rosenburgh & Marrs 2010). The Heather and Grass Burning Code (Defra 2007, see Annex B) notes that "when used with skill and understanding, (burning) can benefit agriculture, game birds and wildlife. But if it is used irresponsibly it creates risks for people, property and the environment." The Code also notes that in some "sensitive areas" burning "may permanently damage the environmental interest of the land". These include upland and other peatlands.

Cool burning

3.10 The Heather and Grass Burning Code (Defra 2007) recommends “quick, cool burns” with the “aim to remove the dwarf shrub canopy but leave behind a proportion of stick and try not to damage the moss or litter layer or expose the bare soil surface.” The recommendation of this approach was new in the 2007 code and followed its adoption in guidance and practice over the previous decade or so. It was promoted on wet heath and blanket bog in the supplement to the Muirburn code in 2001 “to reduce the chance of damage to, and removal of, the protective moss layer” (SEERAD 2001b) and in the Moorland management template for SSSI grouse moors agreed between English Nature and the Moorland Association in 2005 (English Nature & Moorland Association 2005). ‘Cool burning‘ was developed and adopted not just to reduce severity/impact, particularly on the moss and litter layer, and to improve control, but also to try to extend the number of days when burning was possible, ie when the weather was not overly dry and the vegetation remained somewhat damp. This is made possible by the use of the technology of ‘pressurised fuel-assisted burning’ which allows burning when the vegetation is moister.

3.11 A further summary of current Codes of Practice and guidance on burning is given as additional context in Appendix 2.

The current topic review

3.12 To explore the topic more fully, the topic review question was split into a number of sub-questions, or sub-topics, that cover different aspects of burning on upland peatland habitats. Some of these sub-questions differ in nature and can be grouped as described below.

3.13 The first four directly address the overarching question on the effects on biodiversity and ecosystem services by the key interests: a) vegetation composition and structure (including key individual plant species or groups); b) fauna including key groups and species; c) carbon sequestration and storage; and d) water quality and water flow.

3.14 The next two are in effect components of the first four that investigate two particular aspects of the effects across the four interests: (e) differences in the severity, frequency, scale, location and other characteristics of burns; and (f) the interaction of burning and grazing. Evidence in relation to these aspects is mainly drawn from the studies cited in relation to the first four sub-questions and, therefore, a more concise, summary review approach has been adopted so as not to repeat the detail of studies already described under the first four sub-questions.

3.15 The remaining two cover narrower, but separate aspects: (g) the relationship between managed burning and wildfire; and (h) the extent, frequency, practice and type of managed burning. Evidence regarding these aspects is more limited and, especially for the former, is also covered using a more concise, summary review approach.

Treatment of evidence in sub-topic sections and appendices

3.16 The evidence for the eight individual sub-questions is reviewed in subsequent Sections 4-10 and presented in more detail by individual studies in Appendices 3-11. Inevitably, many of the sub-questions are closely related and studies often provide evidence against more than one sub-question. In such cases, the evidence is described in more detail under the topic that it principally relates to, with briefer treatment under other questions tailored to the particular sub-question. References that relate to the same study, particularly from different time intervals during long-term studies or reports/theses and associated published papers on the same study, have generally been assessed and reported together (with the principle reference given first). They are thus regarded as single studies including in the listings in the summaries of evidence statements. The sub-questions have been broken down into component aspects (outcomes or types of study) for presentation of the evidence. This

ensures that related studies are grouped together. Within these component parts, references are given in date order.

4 The effects of managed burning on the maintenance and restoration of the characteristic floristic composition, structure and function of upland peatland habitats

4.1 The full text of sub-question (a) is:

What are the effects of managed burning on the maintenance and restoration of the characteristic floristic composition, structure and function of upland peatland habitats?

Introduction and context

The characteristic vegetation and flora of upland peatlands

Vegetation communities

4.2 Upland peatland habitats are described in paragraphs 3.3 to 3.6. They comprise the UK BAP priority habitats blanket bog, upland flushes, fens and swamps (though some types of this can also occur on mineral soils) and the wet heath component of upland heathland all of which occur on moorland.

4.3 *Trichophorum cespitosum* and bog-mosses such as papillose bog-moss *Sphagnum papillosum,* acute-leaved/red bog-moss *S. tenellum* and S. *capillifolium* are characteristic of blanket bog throughout its UK range. Other species are more characteristic of or more abundant in, certain areas. For example, the higher, drier eastern bogs typically support a higher proportion of hare's tail cotton grass *Eriophorum vaginatum* and bilberry *Vaccinium* myrtilus. Similarly, *Molinia* and bog-myrtle *Myrica gale* are much more widespread and typical on western bogs (Rodwell 1991, Averis *et al.* 2004). *Calluna* also occurs widely, especially in drier situations including on hummocks where it often regenerates through layering (MacDonald *et al.* 1995) and may achieve a 'steady state' through the continuous layering and rejuvenation of stems among the *Sphagnum* carpet (Hobbs 1984). However, the distribution and abundance of many species is not solely reflective of geography, altitude and climate but has been greatly affected by land management, notably drainage, grazing and burning/wildfire (for example, Lindsay 2010).

4.4 Upland flushes, fens and swamps receive water and nutrients from surface and/or groundwater sources as well as rainfall. The soil is generally waterlogged with the water table close to or above the surface for most of the year (Maddock 2008). They include both *soligenous* mires (springs, flushes, valley fens) and topogenous mires (basin, open-water transition and flood-plain fens), as well as certain *Molinia* grasslands and rush pastures. Also excluded are species-poor *Molinia* swards and species-poor or 'weedy' soft-rush *Juncus effusus* swards. Swamps are included except for those forming a fringe less than 5 m wide adjacent to standing waters. Though often too wet to be burned, burning does occur in places particularly in more extensive stands such as some valley mires.

4.5 Wet heath is most commonly found in the wetter north and west and, less-modified forms are dominated by mixtures of cross-leaved heath, *Erica tetralix*, *Trichophorum cespitosum*,

heather and *Molinia*, over an understorey of mosses often including carpets of *Sphagnum* bog-mosses (Maddock 2008). This habitat is distinct from blanket mire which occurs on deeper peat and which usually contains frequent occurrence of *Eriophorum vaginatum* and characteristic mosses.

4.6 The National Vegetation Classification (NVC) provides a systematic and comprehensive catalogue and description of plant communities in Britain (Rodwell *et al.* 1991 et seq.). A list of relevant NVC community types occurring in upland peatlands is given in Table 6. These include bog pool communities that occur on the surface of the blanket peat and a number of communities that can replace characteristic active blanket bog communities on deep peat due to human activity. There are seven main blanket bog NVC mire vegetation types on deep peat: M15-20 and M25. All but three (M16, 17 and 18) represent modified types of blanket bog at least to a degree. There are also three bog pool communities associated with blanket bog: M1, M2 and M3. More severely modified blanket bog vegetation includes dry heath vegetation types, particularly H9 and H12, and some grassland types, particularly U6.

Table 6 Blanket bog (active and non-active) NVC community types on deep blanket peat

NVC community	Characteristics
M1 cow-horn bog-moss *Sphagnum auriculatum* bog pool community	Commonly associated with western blanket bogs.
M2 feathery bog-moss *Sphagnum cuspidatum*/flat-topped bog-moss *S. fallax* bog pool community	Usually occurs within M15, M17 and M18 wet heath/blanket bogs.
M3 *Eriophorum angustifolium* bog pool community	Species-poor community sometimes derived from one of the other bog pool types by management impacts or recolonisation or eroded areas.
M15 *Scirpus cespitosus* – *Erica tetralix* wet heath	Mire community on drier ombrogenous peat, but also as a wet heath community on thinner or transitional peat associated with grazing, burning and drainage of once wetter peats.
M16 *Erica tetralix* – compact bog-moss *Sphagnum compactum* wet heath	Bog community replacing M17 and M19 blanket mire communities associated with heavy grazing, burning, and drying of peat.
M17 *Tricophorum cespitosum* - *Eriophorum* blanket mire	Western and more oceanic blanket mire community characterised by deer-grass, purple moor grass, cotton-grass, heather and cross-leaved heath and *Sphagnum capillifolium* and *S. papillosum.*
M18 *Erica tetralix* – *Sphagnum papillosum* raised and blanket mire.	Similar to above, with typical species being heather, cross-leaved heath and cottongrass on waterlogged peat at lower altitudes. Deer-grass and purple moor grass generally less common.
M19 *Calluna vulgaris* – *Eriophorum vaginatum* blanket mire	Extensive blanket mire type, dominated by heather and cottongrass. Less species rich compared to M17 & M18. One of the drier types of bog which with drainage and regular burning can lead to change to a dry heath community.
M20 *Eriophorum vaginatum* blanket and raised mire	Species poor ombrogenous bog community derived from M19 through intensive management including burning, grazing and drainage. This largely eliminates the characteristic *Sphagnum* species and leads to domination by *Eriophorum vaginatum*. Can also occur in a more 'natural' form at the drier bog edges.
M25 *Molinia caerulea* – *Potentilla erecta* mire	Mire community with overwhelming dominance by purple moor-grass, often associated with areas of peat aeration.

Table continued…

NVC community	Characteristics
NVC dry heath communities on deep peat:	
H9 *Calluna vulgaris* – wavy hair-grass *Deschampsia flexuosa* heath	Blanket bog heavily dominated by heather but with some *D. flexuosa* and purple moor grass on wetter ground associated with areas of burning.
H12 *Calluna vulgaris* – *Vaccinium myrtillus* heath	A heather dominated community on free draining soils associated with burning and grazing.
NVC grassland community on deep peat:	
U6 heath rush *Juncus squarrosus* – sheep's fescue *Festuca ovina* acid grassland	Montane community associated moist peats and encouraged on wetter peats by repeated burning and grazing that lead to domination by rushes and grasses.

4.7 Wet heath comprises M15 and M16 generally on shallow peat, and in degraded form M25 mire and U6 acid grassland (see Table 6 for brief descriptions). Upland flushes; fens and swamps include a wide range of mire and swamp NVC communities: M4-M12, M21, M23a, M25c, M27-M29, M31-M35, M37, M38, S9-S11, S19 and S27.

UK BAP plant and fungi species

4.8 An analysis by Webb *et al.* (2010) showed that of the 98 UK BAP species (including animals, lichens and fungi) that are associated with the uplands in general, only ten (including animals) are associated with blanket bog, although more (23 including animals and fungi) are associated with upland flushes, fens and swamps.

4.9 The ten species associated with blanket bogs occur primarily in northern England. These include three very restricted bryophytes (carrion-moss *Aplodon wormskjoldii*, Baltic bog-moss *Sphagnum balticum* and round-fruited collar-moss *Splachnum vasculosum*). The ten species associated with blanket bog occur primarily in northern England. The bryophytes are highly restricted in distribution, being associated only with nutrient-poor *Sphagnum* - and sedge-dominated wetlands which include: permanently wet conditions; *Sphagnum* , shallow pools and low vegetation cover; high water quality; and low pH (Webb *et al.* 2010), mostly in the north, in England, although *Splachnum vasculosum* is also recorded from Dartmoor, and *Aplodon wormskjoldii* has not been recorded since 1981 and may be extinct in Britain (Atherton *et al.* 2010). Although upland blanket bogs do not support a large diversity of species (UK BAP priority species or otherwise), they are important for the small number of specialist species with localised/very restricted distributions. This factor should be taken into account when comparing this habitat with other upland habitats.

4.10 The twenty three species associated with upland fens, flushes and swamps include: four vascular plants, flat-sedge *Blysmus* compressus, yellow marsh saxifrage *Saxifraga hirculus*, lesser butterfly-orchid *Platanthera bifolia* and the eyebright *Euphrasia rivularis*; four bryophytes, marsh flapwort *Jamesoniella undulifolia*, *Splachnum vasculosum*, fen *notchwort Leiocolea rutheana* and *Thamnobryum angustifolium*; and two fungi, *Urocystis primulicola* and *Armillaria ectypa*. None of these plants or fungi is widespread. Again, in England they are mostly associated with the north, particularly the north-west. The analysis by Webb *et al.* (2010) showed that all these species are associated with open, unshaded habitats and that the majority require high water quality. Additionally, many species require permanently wet habitats dominated by sedges and mosses.

Other species

4.11 There is quite a lot of information published on the autecology of the main peatland plant species, for example in the Biological flora of the British Isles series in the Journal of Ecology, the Ecological Flora database[4] and in the Comparative plant ecology book (Grime *et al.* 1988), though most of this material does not relate specifically to burning and was considered beyond the scope of the current review. However, a number of classifications of plant species have been proposed based on their response to burning. For example, Fraser (1933, [cit. Elliot 1953]) suggested two categories: firstly, caespitose species in which winter buds and food stores are protected by a dead matting of leaf sheaths and last year's leaves (for example, *Eriophorum vaginatum*) and plants whose winter buds and reserve food stores lie beneath the ground (for example, common cottongrass *E. angustifolium*) and which are scarcely affected by fire; and secondly, sub-shrubs and other plants with winter buds at or near the surface of the ground, and with the food reserves stored in the stem and which do not regenerate readily from roots (for example, *Calluna*) which are more readily destroyed by fire. Similarly Rowe (1983, [cit. Lindsay *et al.* 1988]) described four different basic types of plant response to fire: 'resisters', such as *E. vaginatum*, which tolerate and survive fire; 'endurers', such as bearberry *Arctostaphylos uva-ursi* and cowberry *Empetrum nigrum*, which regenerate from below-ground organs; 'evaders', that are able to set seed in peat and germinate after fire; and 'avoiders' such as glittering wood-moss *Hylocomium splendens*, which cannot tolerate fire and rely on long cycles to allow reinvasion and recovery from populations surviving elsewhere. The response of individual species to burning has been further reviewed by several authors, for example, Elliot (1953), Currall (1981), Shaw *et al.* (1996), Tucker (2003) MacDonald (2008), Littlewood *et al.* (2010) and Worrall *et al.* (2011) but much of the information on 'fire-sensitive' species is anecdotal and beyond the scope of this review. Nevertheless, it is clear that plant species are differentially affected by burning.

Studies and evidence for the effects of burning on vegetation composition, structure and function

4.12 Twenty-four individual studies (35 references) and one systematic review, all from the UK and Ireland, provide evidence for sub-question (a) on the effects of burning on vegetation composition and structure. The detailed summarises of the individual studies can be found in Appendix 3. In addition, the findings of other evaluated reviews that relate to the sub-question but are not quoted in the summary and discussion below are also summarised separately in Appendix 5.

Applicability of the evidence for the effects of burning on vegetation composition, structure and function to UK upland peatlands

4.13 The following section reviews the applicability of the evidence from primary studies reviewed to the effects of burning on vegetation composition, structure and function to UK, and in particular, English upland peatlands. It draws on the assessment of the external validity carried out in assessing the studies (see paragraphs 2.14-2.15) and the studies as a whole including such factors as geographical location and how representative the habitat/vegetation type and intervention was of the study area(s) and nationally.

4.14 All of the 24 studies were from the UK & Ireland and all but one from the uplands so are geographically and ecologically applicable to upland peatlands in the British Isles. Six were from Scotland, two from Ireland/N Ireland and one from Wales. Of the remaining 15 in England, seven were from the North Pennines/Northumbria, six from the South

[4] See: **www.ecoflora.co.uk/**

Pennines/Peak District and one each from the Yorkshire Dales, North York Moors and lowland Cumbria (including one study that included sites in both the Peak District and Yorkshire Dales). Thus, the English studies were concentrated in the Pennines (13), with none from some upland areas where upland peatland habitats are of importance including Bowland, the Lake District, Shropshire Hills/Welsh borders and the South West moors. Thus, whilst the studies as a whole are geographically applicable to the main area of English blanket bog in the Pennines, other areas and hence the range of variation in the upland peatland resource as a whole, are not represented. In particular, together with the scarcity of study sites in Wales, western English upland peatlands, with their wetter, milder oceanic-influenced climate are under-represented (though they are represented in some of the Scottish sites).

4.15 Eight of the studies (seven in England) involved experimental burning treatments, although three of these were separate studies using the long-term burning and grazing experiment at Hard Hill [1++] in Moor House NNR in the North Pennines (see Appendix 3) which is the only long-term study that covers multiple burning rotations. The other experiments were established to study particular aspects of post-burn or other treatment response: *Molinia* management (Miles 1971, Ross *et al.* 2003, Marrs *et al.* 2004, [all 1+]) or *Calluna* response (Cotton & Hale 1994 [1-]) but also collected data on wider vegetation composition. Unfortunately, they all only covered the initial (between 2-6 year) post-burn period. Because of the length of time required to actually observe post-burn successional vegetation changes in a single site/stand and the relatively long length of burn rotations, several other studies use 'chronosequences' that substitute space for time in determining the sequence of the post-burn succession from stands of differing ages since burning to look at longer post-burn periods within or between sites (Currall 1981 (pt), Davies & Legg 2001, Harris *et al.* 2006, 2011b [all 2+]; and McFerran *et al.* 1995, Stone 2006 and Burch 2008, 2009 [all 2-]). Such studies make the critical assumption that each site in the sequence differs only in age and that each has traced the same history in both its abiotic and biotic components. This has led to some critiques of the chronosequence method (not just in vegetation monitoring but also in nutrient cycling, productivity and carbon flux studies) that point out some inherent problems and limitations (for example, Collins & Adams 1983; Pickett 1988) and suggest the need to validate or justify the critical assumptions. Johnson & Miyanishi (2008) reviewed evidence from studies that used non-chronosequence methods (such as long-term study of permanent plots) to test the space-for-time substitution in four classic succession studies and showed that empirical evidence invalidated the chronosequence-based sequences inferred in these classic studies. Nevertheless, Walker *et al.* (2010) suggest "that temporal change can be successfully explored through the judicious use of chronosequences especially for communities with long successional trajectories.

4.16 Moor House is not typical in some respects and Gray & Levy (2009) have suggested that extrapolation of results more widely should be done with caution. Lindsay (2010) also expressed some concerns about the effects of trampling from repeated surveys and that the experimental burns of small plots may differ from the larger, normal management burns more typical on moorland, although this is a general issue with experimental burns. However, Adamson & Kahl (2003) suggested that the results from the Hard Hill plots experiment are probably applicable to less-modified, high altitude blanket bog with a restricted grazing regime. Most of the vegetation surveys of the Hard Hill plots have (deliberately) been carried out 6-7 year after the previous burn (for all 5 surveys of the short rotation plots and 3 of the long rotation plots surveys) apart from two, in 1972/73 and 1991, when the 20-year plots were 16-17 year since the last burn. Consequently, overall the main survey data are skewed towards an elapse time of 6-7 year since the last burn, with fewer from 16-17 year and none in other years between the 0-20 year ranges (though the period up to 5 year was at least partially covered in other surveys in 1965 and 1976-80). Thus, whilst the main results enable an assessment of the impact of multiple burn rotations over time, they do not present the full sequence of post-burn recovery up to 20 year. Twenty-five 1 m^2 quadrats per plot were surveyed in 1961 and 20 in all subsequent resurveys except 1965 when only five were surveyed. Plant species abundance was estimated using the 'Domin cover-abundance scale' (Evans & Dahl 1955) in random quadrats 1961 and 1965, and using 'pin frames' with multiple

hits (five pins randomly from 50 positions per m^2, ie 100/plot, 200/treatment, except in 1972 when 10 per m^2) in between 1972/73 and 2001. Given that the plots are 30 m x 30 m (900 m^2), the number/density of pins is low. Especially given the known patchy nature of some species in the plots (for example, Hobbs 1984), this is likely to lead to relatively poor ability to pick up species occurring at low cover (compared, for example, with recording all species and covers in 1 m^2 quadrats as done in the first surveys in 1961 and 1965).

4.17 Most of the studies reviewed are of blanket bog including many modified/severely modified examples often with *Calluna* dominant on deep peat. Some studies relate to wet heath and transitions to humid/dry heath, but none were identified of other upland peatland habitats.

Summary and discussion of the evidence for the effects of burning on vegetation composition, structure and function

4.18 The following section presents a summary and brief discussion of the findings of the review of the evidence for the effects of burning on vegetation composition and structure drawing primarily on the primary studies and systematic review presented above. Where there is clear evidence of specific effects of burning this is summarised as an evidence statement, given in bold, including its strength (see paragraphs 2.23-2.24) and the number of studies that contribute to it and their type and quality categorisation. They relate to burning or to indirect effects through changes in vegetation composition and structure. The statements are ordered from broad to more specific effects.

4.19 There is strong evidence from the majority of studies of vegetation response to burning that managed, rotational burning results, at least for a period, in change in the species composition of blanket bog and upland wet heath vegetation (two [1++], Hard Hill expt. 1961-2001, Stewart *et al.* 2004; four [1+], Taylor & Marks 1971, Ross *et al.* 2003, Marrs *et al.* 2004 and Ward 2007; one [1-], Cotton & Hale 1994; one [2++], Lindsay & Ross 1994; four [2+], Currall 1981, Davies & Legg 2008, Harris *et al.* 2006, 2011b; and six [2-] Elliott 1953, Forrest & Smith 1975, McFerran *et al.* 1995, Hamilton 2000, Stone 2006, Burch 2008,). This results from species (a) changing relative abundance or (b) disappearing and recolonising or potentially colonising (Harris *et al.* 2011b [2+]). The actual species involved vary, it has been suggested in relation to a variety of factors including habitat condition/degree of modification, pre-burn abundance, seed/propagule availability and other factors such as geographical location and site characteristics such as slope and depth of water table (Coulson *et al.* 1992 [4+], Harris *et al.* 2011b [2+]).

4.20 There is strong evidence that burning of blanket bog and wet heath typically leads to an initial period of graminoid dominance, typically of 10-20 year, and at least an initial decline in dwarf shrub cover and diversity (two [1++], Hard Hill expt. 1961-2001 Stewart *et al.* 2004; four [1+], Miles 1971, Ross *et al.* 2003, Marrs *et al.* 2004, and Ward 2007; two [2+], Currall 1981, Harris 2011b; three [2-], Elliott 1953, Forrest & Smith 1975, McFerran *et al.* 1995).

4.21 There is strong evidence that graminoids such as *Molinia* and *Trichophorum* tend to initial dominance after burning, especially in the oceanic, wetter west of the country (three [1+] Miles 1971, Ross *et al.* 2003, Marrs *et al.* 2004; one [2+], Currall 1981; and one [2-], Forrest & Smith 1975; two in Scotland, two in the N Pennines/Northumberland, and one in the Peak District/Yorkshire Dales; and one review: Anderson *et al.* 2006 [4+]) or alternatively *Eriophorum vaginatum*, particularly on blanket bog in the Pennines (one [1++]: Hard Hill 1961-2001; one [1+]: Ward 2007; and one [2+]: Harris *et al.* 2011b). This period of graminoid dominance tends to be longer than typically occurs with dry heath or more severely modified, drier bog (Harris *et al.* 2011b [2+]), probably reflecting the greater competitive advantage of wetland graminoids.

4.22 There is strong evidence that *Calluna* tends to decline during the initial post-burning graminoid-dominant phase, but typically then increases (two [1++], Hard Hill expt. 1961-2001, Stewart *et al.* 2004; three [1+], Ross *et al.* 2003, Marrs *et al.* 2004 and Ward 2007; one [2+],

Currall 1981; and one [2-] McFerran *et al.* 1995), especially on drier sites, though this may take 15-20 year or longer on less modified, wetter blanket bog (one, [1++] Hard Hill expt. 1961-2001; one [1+] Ward *et al.* 2007) and may not occur, for example, with too frequent or severe burning and/or heavy grazing (one study [2+], Currall 1981). *Calluna* may continue to increase in, or maintain high, cover for a considerable period as it grows. On more severely modified, drier sites this may be at the expense of other species so that it becomes overwhelmingly dominant (Harris 2011b [2+]), but on less modified, wetter sites it's stems tend to be constantly reburied by growth of *Sphagnum* and through the rejuvenation of these stems, an uneven aged stand of *Calluna* is produced, the so called 'steady state' where other mire species are well represented (Rawes & Hobbs 1979 and Hobbs 1984 in Hard Hill expt. papers [1++]; and Mowforth & Sydes 1989, Coulson *et al.* 1992, Tucker 2003 and Lindsay 2010, all [4+]).

4.23 There is weak evidence that other dwarf shrubs, especially *Empetrum nigrum*, may decline following burning (one [1-], Cotton & Hale 1994; two [2-], Elliott 1953, Forrest & Smith 1975) and sometimes not recover after more severe fires (Stewart *et al.* 2004 [1++]).

4.24 There is moderate evidence from two related Moor House studies that burning leads to an increase in cloudberry *Rubus chamaemorus* (one [1++], Hard Hill expt. 1961-2001; and one [1+], Taylor & Marks 1971), although another study on the same site reported a decline (Forrest & Smith 1975 [2-]) and re-analysis of data from the main experiment between 1972 and 2001 failed to detect a difference between burning treatments but showed a decline in the grazed, 'unburnt' treatment (Lee *et al.* (2013) as part of the Hard Hill expt. study [1++]). This indicates that that grazing is an important factor that may interact with burning (Taylor & Marks 1971 [1+]).

4.25 There is strong evidence that overall, bryophytes tend to decline initially after burning of blanket bog (one [1++], Hard Hill expt. 1961-2001; two [2+], Currall 1981, Harris 2011b; and two [2-] Forrest & Smith 1975, McFerran *et al.* 1995), although some early-colonising species may relatively quickly become frequent or even abundant (Hard Hill expt. [1++], Burch 2008 [2-]). *Sphagnum* species as a group have shown mixed responses, in some cases increasing in the early post-burn stages (one [1++]: Lee *et al.* (2013) as part of the Hard Hill expt. study; Forest & Smith 1975 and Hamilton 2000, both [2-]), sometimes declining or being killed (Lindsay 1977 [2++], Hamilton 2000 [2-]) and sometimes then increasing or recolonising after varying periods (Miles 1971 Lindsay & Ross 1994 Burch 2008). These differences in response probably reflect a variety of factors including the actual species of *Sphagnum* involved (some being colonisers and other typical bog species slower at colonisation) and the severity of the burn. Nevertheless, high *Sphagnum* cover and diversity is characteristic of less-modified peatland habitats (Lindsay 1977/Lindsay & Ross 1994 [2++]).

4.26 There is moderate evidence (from one long-term study: Lee *et al.* 2013 [1++]) that the composition of blanket bog vegetation can continue to show change more than 80 year after the last burn. These changes included continued increase (growth) in *Calluna*, though not at the expense of other species, and an increase in *Sphagnum* diversity and in some other mosses.

4.27 There is strong evidence that burning is associated with the creation of bare ground, at least at a fine scale (one [1++], Hard Hill expt. 2061-2001; one [1-], Cotton & Hale 1994; one [2+], Currall 1981; and two [2-], Elliott 1953 and McFerran *et al.* 1995) including in both less modified and severely modified blanket bog, and in wet heath (Currall 1981 [2+]). Where recorded cover of bare ground is often relatively low and varies in relation to a variety of factors including burn severity (which may be patchy) and the cover of the bryophyte, lichen and litter ground layer. Areas of bare peat tend to be flatter and lack microtopographical variation (Lindsay 2010 [4+]).

4.28 Few of the studies appear to have specifically recorded variation in structure of vegetation and bog surface microtopography, although there is moderate evidence of relatively flat,

unpatterned bog surfaces resulting from fire and of recovery of hummock-hollow topography following gradual recovery or recolonisation of *Sphagnum* species (Lindsay & Ross 1994 [2++], Hamilton 2000 [2-]). Burning can also result in the formation of tussocks, particularly of *Eriophorum vaginatum* (Shaw *et al.* 1996 [4+]).

4.29 There is relatively little evidence of the effects of differences in burning rotations on peatlands with the only long-term experimental study that has covered multiple rotations of differing lengths (10 and 20-year) being the Hard Hill experiment ([1++]). This provides moderate evidence that differences in frequency of burning affect the vegetation composition and structure of blanket bog. At this site, more frequent burning has promoted dominance of hare's-tail cottongrass, with heather achieving higher cover under the longer rotation.

4.30 Changes in vegetation composition and structure may affect the functioning of the peatland ecosystem and hence have effects on associated ecosystem services which are reviewed in subsequent sub-questions. When interpreted in relation to the characteristic floristic composition, structure and function of upland peatland habitats, overall these vegetation responses to burning, in particular the tendency to dominance of graminoids and/or *Calluna* at different post-burn stages and depending on site conditions, may reduce the chance of maintaining active, functioning peatlands. Similarly, where restoration to favourable condition is an objective for modified, degraded upland peatland habitats, burning may perpetuate dominance of graminoids or *Calluna*.

Research recommendations on the effects of burning on vegetation composition, structure and function

4.31 Assessment of the available evidence suggests that the following areas would benefit from further research. In the Conclusions (Section 12 Research recommendations), these are considered in relation to commonality with recommendations for the other sub-topic questions in order to make identify priority research recommendations across the burning topic:

- The extension of experimental and other monitoring studies of the effects of burning on vegetation to a wider range of sites across the English upland peatland resource, including additional long-term studies to cover multiple rotations across the full length of typical blanket bog burn rotations (for example, 15-25 year) (which are currently restricted to the Hard Hill experiment at Moor House). Such studies should also include the wider range of upland peatland habitats including wet heath and flushes, fens (including valley mires) and swamps.
- Research on post-burn vegetation recovery times in upland peatlands including palaeo-archival studies on vegetation recovery after fire. Research on the effects of burning on the range of characteristic upland peatland species, especially individual *Sphagnum* bog-moss species, including post-burn recovery.
- Research on restoration management, including the potential use of one-off burning and alternative treatments to reduce graminoid and *Calluna* dominance. This is being addressed at two sites in the Pennines and one at Bowland, including the effects on carbon and water, in a current 5-year Defra Environmental Stewardship research project, BD5104[5].
- More detailed and consistent description of habitat/vegetation type, composition (including *Sphagnum* species), structure and condition of sites in peatland research, monitoring and survey studies, not just in vegetation but also in wider studies. This should include more detailed and consistent recording of surface topography/structure.
- Interpretation of the results from existing and new vegetation studies should to be interpreted from an ecological and nature conservation/biodiversity perspective, for

[5] See: **https://sites.google.com/a/york.ac.uk/peatlandesuk/**

example, potentially including aspects of autecology, functional types and associations (for example, through use of Ellenberg and suited species scores), disturbance (CSR etc), habitats and vegetation community types (for example, NVC and Annex 1 habitats), habitat condition (for example, CSM), associated species, structure and function etc.

5 The effects of managed burning on the maintenance and enhancement of the characteristic fauna of upland peatlands

5.1 The full text of sub-question (b) is:

What are the effects of managed burning on the maintenance and enhancement of the characteristic fauna of upland peatlands either directly or indirectly through changes in vegetation composition and structure?

Introduction and context

The fauna of upland peatlands

UK BAP species

5.2 Of the 98 UK BAP species that are associated with the uplands in general, only ten (including plants) are associated with blanket bog UK BAP priority habitat, although more (23 including plants) are associated with the upland flushes, fens and swamps UK BAP priority habitat (Webb *et al.* 2010).

5.3 The ten species associated with blanket bog occur primarily in Northern England. These include: four widespread or localised vertebrates (adder *Vipera berus*, common lizard *Zootoca vivipara*, curlew *Numenius arquata* and black grouse *Tetrao tetrix*); and three localised invertebrates (large heath *Coenonympha tullia* and the money spiders *Semljicola caliginosus* and *Notioscopus sarcinatus*).

5.4 The twenty-three species associated with upland fens, flushes and swamps include twelve invertebrates (sandbowl snail *Quickella arenaria*, round-mouthed whorl snail *Vertigo genesii*, Geyer's whorl snail *Vertigo geyeri*, two money spiders *Semljicola caliginosus* and *Notioscopus sarcinatus*, bog hoverfly *Eristalis cyptarum*, barred green colonel *Odontomyia hydroleon*, narrow-bordered bee hawk-moth *Hemaris tityus*, large heath mountain ringlet *Erebia epiphron*, small pearl-bordered fritillary *Boloria selene*, marsh fritillary *Euphydryas aurinia*); and one bird (curlew). Only three of these species are widespread (curlew, and marsh and small pearl-bordered fritillaries). Again, they are mostly associated with the north of England, particularly the North-West. The analysis by Webb *et al.* (2010) showed that all these species are associated with open, unshaded habitats and that the majority require high water quality. Additionally, many species require permanently wet habitats dominated by sedges and mosses.

Invertebrates

5.5 Relatively little is known about the invertebrate assemblages of moorland and specifically peatland habitats compared, for example to lowland heathland and other lowland habitats. However, there have been a series of surveys and reviews of invertebrates on upland habitats including peatlands, notably by Ratcliffe (1977), Butterfield & Coulson (1985, 1983), Coulson & Butterfield (1986), Coulson (1988), Gardner & Usher (1989), Usher (1992), Holmes (1993), Eyre (2003) and Stone (2006) some of which considered the relationship with management including burning. Coulson (1988) reviewed the structure and importance of invertebrate communities on peatlands and moorland and noted that enchytraeid worms, springtails (Collembola) and mites (Acari) make up the bulk of invertebrates across a wide range of

moorland habitats. Enchytraeid worms also contribute substantially in terms of biomass especially on blanket bog where they can account for over half the standing crop. In terms of insects, true flies (Diptera) and lepidopterans tend to make major contributions to standing crop with Diptera, particularly crane flies, most numerous on blanket bog and other peatlands. Beetles (Coleoptera) and bugs (Hemiptera) are common insect groups on moorland as are spiders (Araneae) and harvestmen (Opiliones). Blanket bog has higher standing crop than dry heath but lower than calcareous grassland (Coulson 1988). In addition to the UK BAP species mentioned above, a range of other nationally rare or scarce invertebrate species occur on moorland habitats including some species associated with degenerate or unmanaged *Calluna* (Eyre *et al.* 2003, Tucker 2003), though some rare species of ground beetles and spiders are associated with short swards resulting from cutting or burning particularly of dry heath (Usher 1992). Though there appears to be little direct experimental evidence of the impacts of burning on invertebrate communities, it is likely to affect invertebrate abundance and diversity in a number of ways, with the immediate effects are catastrophic in terms of direct mortality and reductions in food availability. In most cases, recolonisation is likely to be relatively rapid as many species are relatively mobile (for example, Gardner & Usher 1989). An increase in structural diversity from burning is likely to benefit invertebrate diversity (Grant *et al.* 2012), at least in heathlands and grassland, though the effect may be different in peatlands.

Birds

5.6 The British uplands hold an internationally important assemblage of breeding bird species (Thompson *et al.* 1995b, Ratcliffe & Thompson 1998). Although many of the species are concentrated in Scotland, a number also have significant populations or important outlying range extensions in the English uplands (Brown & Grice 1993). These include several species that occur on upland peatlands, including the Wild Birds Directive 'Annex 1' species: hen harrier *Circus cyaneus*, merlin *Falco columbarius*, golden plover *Pluvialis dominica* and short-eared owl *Asio flammeus*; plus teal *Anas crecca*, red grouse, black grouse, lapwing *Vanellus vanellus*, dunlin *Calidris alpina*, snipe *Gallinago gallinago*, curlew and redshank *Tringa totanus*. Approximately 35% of English moorland is designated as SSSI on the basis of ornithological features and is also included in European Natura 2000 sites as four Special Protection Areas (SPA) (Bowland Fells, North Pennines, North York Moors and South Pennines) (Stroud *et al.* 2001). Whilst Pearce-Higgins *et al.* (2009) note that population trends of upland birds are often poorly known, recent studies and reviews indicate declines in many bird species, particularly waders and some passerines, in the uplands and upland fringe, though some other species are stable and a few increasing (Wotton *et al.* 2002, Fuller *et al.* 2002, Henderson *et al.* 2004, Sim *et al.* 2005, Amar *et al.* 2011).

5.7 Red grouse is a key species in terms of management of upland heathland and peatlands in the north of England with rotational burning management carried out to create a patchwork of *Calluna* of differing age and height to provide young *Calluna* for food and taller stands for shelter and nesting (Hudson & Newborn 1995).

5.8 Burning may impact on birds directly on birds as a result of the potential destruction of nests, eggs and nestlings through fires or indirectly through impacts on habitats and/or other species (Tucker, 2003).

Mammals

5.9 The only large mammal characteristic of moorland habitats is the mountain hare Lepus timidus which, in England, is restricted to the Peak District where it was introduced from the 1880s (Corbett & Harris, 1991). They tend to favour 'heather moorland' including some areas of blanket bog and wet heath. Little appears to be known about the effect of managed burning on hares in the Peak District, although Tucker (2003) and Mallon *et al.* (2003) suggest that from information on their habitat and diet preferences, they are likely to benefit from carefully managed traditional heather moorland burning practices. Rotational burning on grouse moors has been associated with higher mountain hare densities in Scotland (Stoddart & Hewson 1984).

5.10 Small mammals, in particular short-tailed voles *Microtus agrestis*, occur widely on moorland and are of importance as food sources for predatory bird species of conservation importance, especially hen harrier and short-eared owl (Grant *et al.* 2012), but little appears to be know about the effects of burning on their populations.

Studies and evidence for the effects of burning on fauna

5.11 Twenty studies, all but one from the UK, provide evidence on sub-question (c) on the effects of managed burning on the maintenance and enhancement of the characteristic fauna of upland peatlands. The remaining study was from Germany. Nine of the studies related to invertebrates and 11 to birds. The detailed summaries of the individual studies and three evaluated reviews are given in Appendix 4. Some studies that do not specifically consider the direct effects of burning are included when they relate to vegetation composition and structure which may be affected by managed burning. In addition, the findings of other evaluated reviews that relate to the sub-question but are not quoted in the summary and discussion below are also summarised separately in Appendix 5.

Applicability of the evidence for the effects of burning on fauna to UK upland peatlands

5.12 The majority of the studies (18 out of 20) were from, and therefore applicable to, the UK uplands, although most were from Scotland and/or the north of England. Only Tucker (2003) [2+], Moss *et al.* (2005) [2++] and Amar *et al.* (2009) [2++] covered bird data from the UK uplands more widely, although Holmes *et al.* (1993) [2-] sampled ground beetles across Welsh peatlands. Many, especially some of the bird studies do not differentiate peatland from wider moorland habitats which limits their value for the current review, though the some of the bird studies specifically included correlations with burning intensity and/or gamekeeper activity (although it is difficult to separate the effects of burning from predator control). Some of the studies on invertebrates did cover or separate out peatland habitats, although only Curtis & Corrigan (1990) [2-] and Holmes *et al.* (1993) [2-] attempted correlations with burning, although the classification of burning was probably insufficiently precise.

5.13 Many of the studies only provide information on potential indirect effects through burning-induced changes in changes in habitat, especially in vegetation composition and structure, but also potentially other factors such as depth of water table. In addition, there may be direct effects from fire through loss of bird's nests, eggs and young and potentially adult birds and especially reptiles, and invertebrates in various developmental stages.

Summary and discussion of the evidence for the effects of burning on fauna

5.14 The following section presents a summary and brief discussion of the findings of the review of the evidence for the effects of burning on fauna drawing primarily on the primary studies and systematic review presented in Appendix 4. Where there is clear evidence of specific effects of burning this is summarised as an evidence statement, given in bold, including its strength (see paragraphs 2.23-2.24) and the number of studies that contribute to it and their type and quality categorisation. The statements are ordered from broad to more specific effects. Some of the reviews summarised above are also referred to in discussion of the statements.

5.15 No studies were found relating specifically to the effects of burning on upland peatlands on amphibians or reptiles, although one review summarises evidence on generally relatively short-term impacts on reptiles mostly from lowland, and especially heathland, habitats (Glaves *et al.* 2005 [4+]) although adder and common lizard occur on upland peatlands. Similarly, only one study was found relating to burning and mammals and this only related to post-burn grazing of red deer on *Molinia* (rather than any effect on populations, although

Grant *et al.* (2012 [4+]) in relation to field voles *Microtus agrestis* as prey for raptors on moorland in general noted that "field voles are also important prey for hen harriers, and the main prey for short-eared owls (Village 1987, Redpath *et al.* 2002). There appear to be no studies examining the effects of rotational muirburn on vole densities, but these are highest in grassy habitats with little or no grazing (Hewson 1982, Evans *et al.* 2006, Wheeler 2008). Consequently, rotational muirburn seems unlikely to benefit vole abundance because it will promote *Calluna* cover in most situations, and Hope *et al.* (1996) suggested that vole increases were most likely where reductions in sheep and red deer grazing impacts were associated with absent or infrequent burning. However, on blanket bog benefits of burning for voles could arise via the resultant increase in hair's-tail cottongrass, provided grazing levels are low (Wheeler 2008)."

5.16 No primary evidence was found on any specific relationship between burning and the other UK BAP species on upland peatland habitats. However, other reviews indicate that some other animals, notably the two UK BAP reptile species associated with blanket bog, adder and common lizard, may be affected by burning (Glaves *et al.* 2005 [4+]).

Invertebrates

5.17 There are relatively few studies on the impacts of managed burning on the invertebrates of peatlands in general and upland peatlands in particular. None have directly studied the impacts of burning, with most considering burning amongst other management and environmental explanatory variables to explain differences in invertebrate community composition as sampled in site comparison studies, usually, in the case of burning, with a simple classification (for example, recently burned, burned but not recently and unburned). Most studies relate to beetles, especially ground beetles, and spiders, though some other groups are included in a few studies, and some consider the invertebrate community as a whole. One study specifically covered the UK BAP species, large heath (Dennis & Eales 1997, 1999 [2+]).

5.18 There is strong evidence: one [2++]: Hochkirch & Frauke 2007; two [2+]: Eyre *et al.* 2003 and McFerran *et al.* 1995; and five [2-]: Coulson 1988, Curtis & Corrigan 1990, Usher 1992, Holmes 1993 and Stone 2006, mostly involving multiple sites, and one a national sample (from Wales), that burning indirectly influences the invertebrate community composition of upland peatland habitats, typically benefiting open-ground species such as ground beetles and surface-active spiders. Many of the studies indicated an increase in overall species-richness and suggest that this occurs through increases in structural diversity at a relatively fine scale in terms of sward height, especially of *Calluna*, and the presence of open and short swards, resulting from patchwork burning. Thus, conditions are provided for both open ground species and for species that favour taller vegetation such as some web-spinning spiders. Many of the studies were carried out on modified upland peatlands and hence not all of the invertebrate species and assemblages are necessarily characteristic peatland species associated with less modified, functioning, high quality upland peatland habitats.

5.19 However, the majority of studies appear to have been carried out on modified blanket bog, or in some cases wet heath, habitats that tend to dominance by either graminoids or *Calluna*. In less modified or unmodified blanket bog and other peatlands, variation in surface topography develops from the hummock-hollow growth forms of Sphagna and conditions are wetter which may favour a different invertebrate assemblage of more characteristic peatland/wetland species.

5.20 There is weak evidence (from two linked studies: Dennis & Eales 1997, 1999 [2+]) across most known and potential sites in the species' main range in England in Northumberland, that too frequent burning is likely to render peatland sites less suitable or unsuitable for the large heath butterfly (a UK BAP species), but that occasional burning may be beneficial perhaps in favouring the larval foodplant, *Eriophorum vaginatum*, and in reversing succession on at least some drier sites. Although the large heath merits priority species status under the UK BAP

because of its long-term distribution decline, recording during the 2000-04 period suggests that recent losses may not be quite as severe as feared; targeted surveys in Northumberland found little evidence of decline over the past decade and previously unknown colonies were discovered in the North York Moors (Fox *et al.* 2006).

5.21 No evidence was identified on any relationship between burning and the other UK BAP invertebrate species associated with upland peatland habitats.

Breeding birds

5.22 There is strong evidence (one [2++], Amar *et al.* 2009; four [2+], Haworth & Thompson 1990, Smith *et al.* 2001,Tharme *et al.* 2001 and Pierce-Higgins & Grant 2006; and one [2-], Daplyn & Ewald 2006) of correlations between moorland habitat types, and particularly their vegetation composition and structure, and densities of some moorland breeding birds, particularly waders, although in relatively few studies has this been related directly to peatlands rather than moorland in general, or specifically to burning practice (which can be difficult to separate from wider gamekeepering activity including predator control, but see below). Nevertheless, burning on upland peatlands is likely to be an important factor in influencing changes in vegetation composition and especially structure (see sub-question (a)) which may for example affect suitability for nesting and feeding (for example, Grant *et al.* 2012 [4+]). Certain species tend to be associated with particular moorland vegetation characteristics that may be influenced by burning: red grouse and stonechat associated with increasing *Calluna* cover; snipe and curlew with heterogeneity in vegetation structure; golden plover and skylark with short vegetation; waders with wet conditions; whinchat with dense vegetation; stonechat with tall vegetation; and meadow pipit with grass-heather mixes.

5.23 There is strong evidence of correlations between burning and/or predator control intensity (typically associated with grouse moors) and densities of some moorland breeding birds (five [2+]: Picozzi 1968, Haworth & Thompson 1990, Smith *et al.* 2001, Tharme *et al.* 2001 and Pierce-Higgins & Grant 2006; and one [2-]: Daplyn & Ewald 2006). Of these, two studies showed higher densities of red grouse, golden plover (both [2+]: Picozzi 1968, Tharme *et al.* 2001) and curlew (one [2+]: Haworth & Thompson 1990 [2+]; and one [2-]: Daplyn & Ewald 2006) with increased burning/predator control; and single studies higher densities in lapwing (Daplyn & Ewald 2006 [2-]), redshank (Haworth & Thompson 1990 [2+]) and ring ouzel (Daplyn & Ewald 2006 [2-]). Two studies (both [2+]: Smith *et al.* 2001, Tharme *et al.* 2001) showed lower densities of meadow pipit and one (Daplyn & Ewald 2006 [2-]) showed lower densities of skylark, wheatear and twite with increasing intensity of burning/predator control. However, in these studies most (other) species did not show significant correlations with burning/predator control. One study (Fletcher *et al.* (2010) [1+]) showed moderate evidence of an increase in breeding success and numbers of lapwing, golden plover, curlew and red grouse and breeding success of meadow pipit in response to legal predator control, indicating that such control contributes to the increases shown by some species on grouse moors in other studies in addition to any burning effects.

5.24 There is weak evidence (from one study: Smith *et al.* 2001 [2+]) of a correlation between burning and/or predator control intensity and overall diversity of moorland breeding birds, although the same study showed no relationship with species richness.

5.25 Only one study (Amar *et al.* 2009 [2++]) looked in detail at changes in numbers of breeding birds (for five waders) in relation to burning; this showed moderate evidence of greater declines in golden plover under more intensive (rather than less intensive) burning management and that curlew and lapwing declined more on '*Calluna*-dominated' plots than on 'bog' plots.

5.26 One study (Moss *et al.* 2005 [2++]) based on two separate large national datasets, and two studies/reviews (Tucker 2003 [2+]; Ratcliffe 1990 [4+]) of smaller, preliminary datasets, show strong evidence of overlap in the first half of April between the dates of first egg-laying of

some moorland bird species and the legal burning season (which closes on 15 April in the uplands in England). In all but one case, this comprised the minority of first nest attempts (with nine species having more than 10% of and first egg laying attempts before 15 April). Six of these species were potentially vulnerable ground or low vegetation nesting species of which short-eared owl and stonechat were perhaps the most at risk with the others: lapwing, snipe, golden plover and redshank. However, these results only indicate the potential vulnerability of moorland birds to burning; they do not show what proportion of nests is actually affected. The true vulnerability depends on a variety of factors including the choice of nest site in relation to the types of moorland vegetation that are burnt, the proportion of nesting habitat actually subject to burning in any year, the frequency (rotation) with which moors are subject to controlled burns and the effect of burning on nesting attempts/success (including re-nesting). The impact on populations will also depend on the proportion of the population nesting on moorland and in other habitats that may be subject to burning (Moss *et al.* 2005 [2++], Glaves *et al.* 2005 [4+] and Grant *et al.* 2012 [4+]). The study also showed moderate evidence of earlier nesting over time for eight species, which may increase the proportion of first nest attempts by mid-April in future.

5.27 Some of these findings relate to one UK BAP species, curlew, and two Wild Bird Directive Annex 1 species, golden plover and short-eared owl.

General

5.28 No primary evidence was found on any specific relationship between burning and the other UK BAP species on upland peatland habitats. However, other reviews indicate that some other animals, notably the two UK BAP reptile species associated with blanket bog, adder and common lizard, may be affected at least in the short-term by burning (Glaves *et al.* 2005 [4+]).

Research recommendations on the effects of burning on fauna

- Research on the effects of burning on key characteristic blanket bog species of fauna particularly invertebrates and birds (including food availability, for example, Tipulids as wader food).
- Grouse moors (mapping) for correlation studies particularly with breeding bird survey data.
- Further examination of data on bird nesting dates and breeding success in relation to burning (for example, from Nest Record Cards, vulnerability/risk from burning (especially short-eared owl and stonechat) and pre-nesting activity timing).

6 The effects of managed burning of upland peatlands on carbon sequestration

6.1 The full text of sub-topic question (c) is:

What are the effects of managed burning of upland peatlands on carbon sequestration, either directly or indirectly through changes in vegetation composition and structure?

Introduction and context

6.2 Peatlands have great capacity for long-term carbon storage. When undamaged they are waterlogged, which slows down decomposition and enables semi-decomposed plant remains to be laid down as peat. Carbon is removed from the atmosphere into the plant tissue by photosynthesis and it is then stored in the dead plant remains often over millennia, as a thick layer of peat (Bain *et al.* 2011). *Sphagnum* mosses are the main peat-forming species and hence keystone species for providing a range of ecosystem services. UK peatlands hold major stocks of terrestrial carbon, perhaps 3.2 billion tonnes, although more than 70% of this occurs in Scotland (Smith *et al.* 2007; Bain *et al.* 2011). Increasing understanding of peatland functioning is illuminating their role in delivering a broad range of ecosystem services which comprise their social and economic value to society (Defra 2007, 2010). They are important not only as carbon stores but also as major sources of drinking water and as determinants of water flow and hence downstream flood risk (Smith *et al.* 2007, Holden *et al.* 2007) (these water-related functions are dealt with in the subsequent Section 7, although fluvial fluxes are also a component of the overall carbon balance). However, the way that peatlands function is fundamentally affected by their condition, which is itself the product of land management and other environmental pressures, and can often be indicated by land cover or vegetation. Thus, the effects of managed burning on carbon sequestration of upland peatlands is an important and much debated issue.

The studies and evidence for the effects of burning on carbon sequestration

6.3 Eighteen studies provide evidence on sub-question (c). The majority were carried out in, or used data from, the United Kingdom (15), including 14 in, or including data from, England. The remaining three were from Finland, Belarus and the USA (Alaska). Nine of the studies were carried out at (or partially at) Moor House NNR, including four at the Hard Hill burning experiment plots, which are regarded as separate studies below. Other UK studies comprised sites in the Peak District (4), Yorkshire Dales (1), North York Moors (1), Northumberland (1) and NE Scotland (1), plus one systematic survey of UK upland blanket bog catchments (for 'peat pipes'). Some studies that don't specifically consider the direct effects of burning are included when they relate to vegetation composition and structure which may be affected by managed burning. The detailed summarises of the individual studies can be found in Appendix 6 along with the findings of two evaluated reviews that relate to the sub-question.

Applicability of the evidence for the effects of burning on carbon sequestration to UK upland peatlands

6.4 Fifteen of the 18 studies were from the UK, with the majority from or including England (14) and only one from Scotland. However, all of the English studies were from the Pennines and

Northumberland including nine at least partly involving data from Moor House NNR in the North Pennines (also see applicability under sub-question a regarding the Hard Hill experiment and Moor House). All the UK studies probably relate to blanket bog (although it is not always absolutely clear from site descriptions), mostly in modified condition. Whilst the Pennines hold the largest extent of blanket bog in England, both blanket bog in other areas and other peatland habitats more generally are under-represented. Nevertheless the results are applicable to the UK, at least for the Pennines.

Summary and discussion of the effects of burning on carbon sequestration

6.5 The following section presents a summary and brief discussion of the findings of the review of the evidence for the effects of burning on carbon sequestration and storage drawing primarily from the primary studies presented in Appendix 4. Where there is clear evidence of specific effects of burning this is summarised as an evidence statement, given in bold, including its strength (see paragraphs 2.23-2.24) and the number of studies that contribute to it and their type and quality categorization (ordered by type/quality class and date). The statements are ordered from broad to more specific effects. The two UK reviews summarised above are also referred to in discussion of the statements. In a review of UK carbon flux research, Gray & Levy (2009) ([4+]) concluded that "there is insufficient data at present for firm conclusions to be drawn on the effects of fire on the carbon cycle of peatlands or to sufficiently parameterise local or national models."

6.6 There is strong evidence (from 11 studies): one [1++]: Ward *et al*. 2007; two [1+]: Garnett *et al*. 2000 and Orwin & Ostle 2012; one [2++]: Clay *et al*. 2010b; four [2+]: Kinako & Gimmingham 1980, Clay & Worrall 2011, Harris *et al*. 2011a, Allen *et al*. 2013; one [2-]: Farage *et al*. 2009 ; and two modelling studies: one [2+]: Worrall *et al*. 2010a, and one [2-]: Couwenberg *et al*. 2011 that managed burning affects various components of the carbon cycle of peatlands. This includes the following individual aspects.

6.7 There is moderate evidence from two separate studies at the Hard Hill plots at Moor House NNR that (experimental) burning reduces peat accumulation (Garnett *et al.* 2000 [1+]) and reduces above and below ground carbon storage (Ward *et al.* 2007 [1++]) compared to no burning.

6.8 There are few studies of gaseous exchange on sites under managed burning. However, there is moderate evidence (from one of the Hard Hill plots studies: Ward *et al.* 2007 [1++]) of increases in both gross CO2 fluxes of respiration and photosynthesis in (experimentally) burned (10- year) and grazed treatments relative to no burning (controls not burned since 1954).

6.9 There is moderate evidence (Clay & Worrall 2011 and Allen *et al.* 2013 [both 2+]; and Farage *et al.* 2009 [2-]) of carbon losses through fuel consumption during burning and in conversion to char.

6.10 There is moderate evidence (from two sites in one study: Kinako & Gimingham 1980 [2+]) that managed burning can result in erosion and reduction in the level of the soil surface, although there was only weak evidence of a relationship with slope. Erosion reached a maximum within eight months of a burn, with stability restored after 15-20 months. In another study (Shelter *et al.* 2008 [2+]) of wildfires in north America, burning significantly reduced organic soil depth organic matter stocks, although sites dominated by Sphagna had more than twofold greater soil organic matter stock remaining.

6.11 Only relatively recently have attempts been made to estimate complete carbon budgets that draw on the findings for the various components considered above (and colouration/DOC and POC production considered under sub-question d) and consider the overall impacts of burning, often using modelling approaches. Five studies (two [2++]: Garnett *et al*. 2001, Clay

et al. 2010b; [three] [2+]: Worrall *et al*. 2010a and Clay & Worrall 2011 ; and one [2-]: Farage *et al*. 2009) provide carbon budgets either for individual sites or catchments, or across sites or catchments, four relating to managed/experimental burning. The results reported show inconsistent evidence with predictions of both positive and negative overall effects of burning, although they provide strong evidence that burning affects the processes controlling carbon budgets of upland peatlands. One model (Worrall *et al*. 2010a [2+]) derived from a meta-analysis combining data across studies suggested that there would be carbon benefits from not burning upland peatlands. The vegetation composition/condition of the study sites is not always clear from the studies, nor whether the comparisons with unburned sites relate to sites of similar composition/condition or to less modified, active blanket bog. The former is the case in at least some of the studies.

6.12 The apparent differences in response between carbon budget estimates may at least in part reflect differences between sites, for example in starting vegetation composition (including the degree of modification or degradation of unburned stands used as controls in the comparisons) and the severity of burns. The models may also be sensitive to changes in the estimates for some parameters that will also vary in the amount and accuracy of the data that they are derived from.

Evidence gaps and research recommendations on the effects of burning on carbon sequestration

- Away from Moor House NNR, there are relatively few studies of aspects of the upland peatland carbon cycle across the range of English upland peatlands and involving actual management (rather than experimental) burns and, therefore, research and monitoring needs to be extended to new sites across the resource.
- The Hard Hill plots at Moor House NNR is the only long-term study across multiple burn cycles with other studies generally not covering the full length of typical blanket bog burn rotations (for example, 15-25 year). There would be benefit from establishing additional long-term studies.
- There is a relative lack of information on gaseous exchange; and on char production and ‘black carbon’.
- There is a need to better and more consistently describe the characteristics of carbon study sites, for example, in terms of degree of modification, including vegetation composition (including *Sphagnum* species) and structure, and condition.

7 The effects of managed burning of upland peatlands on water quality and flow

7.1 The full text of sub-topic question (d) is:

What are the effects of managed burning of upland peatlands on water quality (including colouration, release of metals and other pollutants and aquatic biodiversity) and water flow (including downstream flood risk), either directly or indirectly through changes in vegetation composition and structure?

Introduction and context

7.2 The problem of increasing water colouration and dissolved organic carbon (DOC) in watercourses draining upland peatland catchments is a water quality issue and costly problem for water companies and also represents a loss of carbon from the system (Worrall *et al.* 2009, Holden *et al.* 2012). DOC in has been increasing in recent decades in watercourses on a wide geographical scale across northern Europe and North America and across moorland and forest habitats (Monteith & Evans 2000, Freeman *et al.* 2001). A range of potential drivers of these trends have been suggested, including temperature, rainfall, acid deposition, land-use, nitrogen and CO_2 enrichment, with Evans *et al.* (2005) suggesting that the increase may be a response to a combination of declining acid deposition and rising temperatures. However, more recent studies have indicated that local moorland land management practices, in particular rotational burning, may be important drivers (Yallop *et al.* 2010, Holden *et al.* 2012). Losses of particulate organic carbon (POC) also occur from peatlands particularly through erosion of the peat surface, though it tends to be a more minor component of fluvial fluxes (Holden *et al.* 2007, Smith *et al.* 2007). Land management practices on peatlands may have effects on water flow and flood peaks downstream (Grayson *et al.* 2010), and may also affect water flow indirectly through changes in vegetation (Holden *et al.* 2008).

7.3 The effects of burning upland peatlands on aquatic invertebrates are included under this sub-question as an aspect of water quality, but are also relevant to sub-question (b) on fauna.

The studies and evidence for the effects of burning on water quality and flow

7.4 Twenty-two studies all from the England (though two overlapped with Scotland) provide evidence on the effects of burning on water quality and flow. The English sites were almost equally split between the southern Pennines (including the Peak District and Yorkshire Dales) and North Pennines, with one study including the North York Moors. A few studies are included that did not specifically consider the direct effects of burning but were relevant to the potential indirect effects through changes in vegetation composition and structure. The detailed summarises of the individual studies can be found in Appendix 7 along with the findings of two evaluated reviews that relate to the sub-question.

Applicability of the evidence on the effects of burning on water quality and flow to UK upland peatlands

7.5 All of the 22 studies were from England and Scotland and are therefore applicable to, the UK uplands. However, all but one of the English studies were from the Pennines or

Northumberland with the other including sites in the North York Moors. All probably relate to blanket bog (though it is not always clear from site descriptions). Whilst the Pennines hold the largest extent of blanket bog in England, both blanket bog in other areas and other peatland habitats more generally are under-represented.

7.6 Some of the studies only provide information on potential indirect effects through burning-induced changes in habitat, especially in vegetation composition and structure, but also potentially other factors such as depth of water table.

Summary and discussion of the effects of burning on water quality and flow

7.7 The following section presents a summary and brief discussion of the findings of the review of the evidence for the effects of burning on water quality and flow drawing primarily on the primary studies and systematic review presented in Appendix 6. Where there is clear evidence of specific effects of burning this is summarised as an evidence statement, given in bold, including its strength (see paragraphs 2.23-2.24) and the number of studies that contribute to it and their type and quality categorisation. The statements are ordered from broad to more specific effects. The reviews summarised above are also referred to in discussion of the statements.

Water colouration and DOC

7.8 There is strong evidence (from five catchment studies and one model: three [2++]: Yallop & Clutterbuck 2009, Clutterbuck & Yallop 2010, Yallop *et al.* 2010; two [2+]: Yallop *et al.* 2008, Grayson *et al.* 2012; and one [2-] Mitchell & McDonald 1995/McDonald *et al.* 1991) that recent burning on deep peat is correlated with an increase in water colouration and/or DOC in watercourses at the catchment scale. In addition, Beharry-Borg (2009) [2+] found an association between the proportion of catchment area burnt and a change in the composition of DOC. However, such effects were not found in two other catchment studies (O'Brien *et al.* 2005 and Chapman *et al.* 2010, both [2-]); this may be related, at least in part, to relatively low sample sizes, not controlling for other factors such as the area of deep peat, imprecision in mapping areas burnt and not separating out recent burns (Yallop *et al.* 2011, Holden *et al.* 2012 [2++]).

7.9 There is moderate evidence (Beharry-Borg 2009 [2+] and Armstrong *et al.* 2012 [3+]) that the area of *Calluna*-dominated vegetation on deep peat is correlated with an increase in water colouration and/or DOC, in soil water in one case and in watercourses draining peatland catchments in the other. Given that on blanket peat *Calluna* is most likely to be burnt, this might be related to burning and/or other factors associated with *Calluna*, such as possibly deeper water tables (Worrall *et al.* 2007, Clay *et al.* 2009 [1++]) and increased peat pipes (Holden *et al.* 2011, 2012 [2++]).

7.10 There is weak evidence from laboratory studies that burning is associated with increased pH and an increase in water colouration (MacDonald *et al.* 1991 and Miller 2008 both [2++]; and Allen 1964 [2+]). These findings may be related as pH controls the solubility of DOC; the higher the pH the greater the solubility of DOC (Holden *et al.* 2011/2012 [2++]). Increases in pH with burning have also been found in laboratory studies on other soil types (Holden *et al.* 2012 [2++]).

7.11 However, small plot- or stand-scale studies of water colouration and/or DOC in relation to burning have apparently only been carried out in soil water in two studies at three upland peatland sites (Clay *et al.* 2012 [2+] and Worrall *et al.* 2010 [2-]) in addition to two separate studies at Hard Hill plots (Worrall *et al.* 2007/Clay *et al.* 2009b and Ward *et al.* 2007, both [1+]), and have shown inconsistent evidence. Little difference between treatments was detected in the Hard Hill plots at the end of the (10-year) burn cycle in both studies, although there was a short-lived peak in both water colouration and DOC in the period following a new

burn monitored in one of the studies. In the other two stand-scale studies of burn chronosequences, one showed a decline in colouration and/or DOC with burning (and grazing) and the other an increase in colour in the first four years after a burn, but no consistent trend in DOC. These findings are not necessarily inconsistent with those from laboratory- and catchment-scale studies as the effect appears to occur in the initial period after new burns (rather than over the whole burn cycle that may be studied in experimental plots) and tends to only occur in the upper layer of peat so may not be picked up by sampling that includes deeper water. Some of these studies have shown some differences in responses between water coloration and DOC (Clay *et al.* 2009b [1+] and Clay *et al.* 2012 [2+]).

7.12 Overall, based on the studies and statements above and supported by a recent, in depth critical synthesis (one study, two references: Holden *et al.* 2011, 2012 [2++]), there is strong evidence that moorland burning results in increased water colouration and/or dissolved organic carbon (DOC) (which is also relevant to sub-question c).

Soil and water chemistry

7.13 There is weak evidence (Worrall & Adamson 2008/Clay *et al.* 2010a [1+]) of differences in concentrations of chemical entities in soil water after a burn, with aluminium, iron and sodium increasing and calcium, chlorine and bromine, and pH declining in soil water, although runoff water showed different trends.

Watercourse aquatic invertebrates

7.14 There is moderate evidence (Asprey 2012 and Ramchunder *et al.* 2013/Ramchunder 2010/Brown *et al.* 2010 [both 2++]; and Ramchunder *et al.* 2009 [2+]) that burning is correlated with changes in the diversity and composition of aquatic invertebrate assemblages in watercourses draining upland peatland catchments. These changes reflect declines in certain groups, especially mayflies and stoneflies and increases in flies.

Hydrology and water flow

7.15 There is moderate evidence (Worrall *et al.* 2007/Clay *et al.* 2009a [1+]; and Worrall *et al.* 2010 [2-]) of shallower water tables initially after burning. The mechanism for this is uncertain, but it has been suggested that it may reflect reduced evapotranspiration or simply a greater proportion of rainfall hitting the surface after removal of the vegetation particularly *Calluna* (Worrall *et al.* 2007/Clay *et al.* 2009a [1+]).

7.16 There is moderate evidence (Clay *et al.* 2009a [1+]; and Clay *et al.* 2012 [2+]) that surface runoff occurs more frequently after recent burning. The mechanisms for this are also uncertain, but may be related to higher water tables, more rainfall reaching the surface after removal of vegetation, the generation of hydrophobic and physiochemical changes in the soil (Clay *et al.* 2009a [1+]) or the production of fine sediment and ash which block macropores (Holden *et al.* in press [1+]).

7.17 No evidence was identified specifically on the impact of burning on the risk of downstream flood events. Pattison & Lane (2011) [4+] explored the problems associated with extrapolation to the catchment scale from small-scale or sub-catchment studies in which land-use management is observed to affect local hydrology, especially if such data are then used to formulate catchment or flood risk management policies. They show how flood response to land-use is not only highly site specific, but also inextricably linked to climatic patterns.

Research recommendations on the effects of burning on water quality and flow

- Extension of plot/stand-scale hydrology and soil water studies to other sites across the English upland peatland resource. This has been done in part through the nearly completed EMBER project, see earlier.
- Extension of studies on aquatic invertebrates more widely across the English uplands. Interpretation of changes in community composition in terms of water quality and biodiversity, possibly including as food for predators (for example, fish and birds such as dipper).
- Studies of the effects of differences in the characteristics of burn patches such size, shape, location (for example, in relation to slope, watercourses etc), distribution etc on water quality, chemistry and flow in peatland watercourses.

8 The effects of differences in the severity, frequency, scale, location and other characteristics of burns on upland peatland biodiversity, carbon and water

8.1 The full text for sub-topic question (e) is:

How do differences in the severity, frequency, scale, location and other characteristics of burns (including 'cool burns') affect upland peatland biodiversity, carbon and water?

Introduction and context

8.2 The pattern, frequency with which fires burn, how hot or intense they are and how damaging or severe they are termed a 'fire regime' (Legg & Davies 2009). Fire intensity refers to the rate at which a fire produces heat at the flaming front expressed in terms of temperature or heat yield. Fire severity, on the other hand, describes the immediate effects of fire on vegetation, litter, or soils. Unlike fire intensity, fire severity "cannot be expressed as a single quantitative measure that relates to resource impact" (Robichaud *et al.* 2000). Instead, fires are typically ranked from low to high severity based on the post-fire appearance of soil, litter, vegetation, or other resource of interest. Burn severity depends not only on the amount of heat generated along the flaming front of a fire (ie intensity) but also on the duration of the burn. Duration is a function of the fire's rate of spread and subsequent 'smoldering' time. Both depend on weather conditions and the nature of the vegetation fuels. Rate of spread is additionally influenced by topography and wind speed. While a fast-moving, wind-driven fire may be intense, a long-lasting fire that just creeps along in the forest underbrush could transfer more total heat to plant tissue or soil. In this way, a slow-moving, low-intensity fire could have much more severe and complex effects on something like forest soil than a faster-moving, higher-intensity fire in the same vegetation.

8.3 The concept of 'cool burns' was introduced about ten years ago and especially from around 2005-2007 when it was introduced in the then new Heather and grass burning code (Defra 2007, see also Appendix 2). The aim is to achieve quick, cool burns which remove the dwarf-shrub canopy but leave behind a proportion of 'stick' and avoid damage to the moss, litter or lichen ground layer or expose bare soil, ie are less severe. It is made possible by the use of the technology of 'pressurised fuel-assisted burning' which allows burning when the vegetation is moister than was possible before. The practice is thought to have been relatively widely adopted, although there is little detailed evidence on the extent of its use (and that available evidence is reviewed under sub-question h). Although relatively few of the evaluated studies specifically mention cool burning, it is likely that it has been used in much of the research carried out over the last ten years that involved normal management burns. Experimental burns tend to be much smaller and hence well controlled, so probably represent or are similar in effect to cool burns.

The studies and evidence for the impact of differing characteristics of burns

8.4 Five studies, all but one from the UK, with the other a continental laboratory study, provides evidence for sub-question (e). Of the UK studies, three were from Scotland and one from England (the last comprising a single site in the Peak District). The summary also draws on the findings from other studies described under sub-questions (a-d).

8.5 The detailed summaries of the individual studies and one evaluated review are given in Appendix 8.

Applicability of the evidence for the impact of differing characteristics of burns to UK upland peatlands

8.6 Of the UK studies included, three were from Scotland and one from England (the last comprising a single site in the Peak District). Thus, they are limited in distribution in relation to the extent of upland peatland in the UK and especially England. A full review however of the more extensive literature of fire characteristics is beyond the scope of this review, in part because most studies don't relate directly to differences in effects on the four aspects of upland peatlands reviewed in the main sub-questions (a-d) and because many relate to *Calluna* as perhaps the most important flammable species and have been carried out on dry heath (or soil type is not specified) rather than on peatlands. Nevertheless, the following references were identified as being relevant to consideration of the effects of burning on upland peatlands, though most do relate to *Calluna*-dominated vegetation, some of which is probably not on deep peat.

Summary and discussion of the effect of differing characteristics of burns on biodiversity carbon and water

8.7 The following section presents a summary and brief discussion of the findings of the topic review of the evidence for the effects of differing characteristics of burns on biodiversity, carbon and water drawing primarily on the primary studies and systematic review presented first above. Where there is clear evidence of specific effects of burning this is summarised as an evidence statement, given in bold, including its strength (see paragraphs 2.23-2.24) and the number of studies that contribute to it and their type and quality categorisation. The statements are ordered from broad to more specific effects.

8.8 Few studies were identified that related differences in the characteristics of burns directly to differences in effects on the four aspects of upland peatlands reviewed in the main sub-questions (a-d). This in part relates to the difficulties in measuring both the intensity and severity of burns and that these aspects tend not be recorded in studies on impacts even in basic terms, such as burning techniques used, for example, pressurised fuel-assisted or 'cool burns'. Some studies did not even make distinctions between whether fires were managed burns or wildfires.

8.9 Thus, as well as the studies described above, the following section draws on findings on the effects of burning on biodiversity and ecosystem services presented under sub-questions (a-d) and on evidence for the extent, frequency, practice and type of managed burning on upland peatlands presented under sub-question (h).

8.10 There is strong evidence (one [1+]: Benscoter *et al.* 2011; and three [2+]: Davies (2005), Davies *et al.* 2010 and Davies & Legg 2011; and one review: Legg & Davies 2009 [4+]) that moisture content, vegetation type and phenology, recent weather and human factors are important factors the ignition of fires.

8.11 There is moderate evidence (Davies (2005), Davies *et al.* 2010a and Davies & Legg 2011 [all 2+]; and one review: Legg & Davies 2009 [4+]) that fuel load and structure are critical factors in fire behaviour, particularly in fireline intensity (heat output per unit length of fire front) and rate of spread, although residence time and depth of penetration of lethal temperatures into the soil are also important in determining severity of effect, but are less well understood. Harris *et al.* (2011a) [2+] showed wide variation in the percentage of biomass removed in 'cool' management burns.

8.12 Little evidence was identified concerning the types of burning practice taking place in the English uplands in general and specifically on deep peat, including on the extent to which 'cool burning' is practised. There is, however, moderate evidence (from two national sample surveys: Critchley *et al.* 2011a, b, both [2++]) that burns into the bryophyte and lichen layer occur in a proportion of cases on blanket bog (11% of all samples) and wet heath (17% of all samples).

8.13 There is evidence that, in addition to initial fire severity, pre-fire vegetation composition (influenced by past fire history amongst other things) is an important factor influencing post-fire recovery (see evidence described under sub-question a).

8.14 The main evidence for the effects of frequency of burning comes from studies from the only long-term burning experiment which involves multiple (10- and 20-year) rotations, at Hard Hill in Moor House NNR described under sub-questions (a, c and d). Even then, the only long-term monitoring relates to vegetation, with the studies on carbon sequestration and hydrology/water quality more recent, although they have shown some differences between burn treatments. Nevertheless, there is moderate evidence (from one study, the Hard Hill experiment [1++]; described under sub-question a) that frequency of burning (ie rotation or return period) affects vegetation composition and structure on blanket bog. At this site, more frequent burning has promoted dominance of hare's-tail cottongrass, with heather achieving higher cover under the longer rotation.

8.15 Some of the catchment-scale studies are relevant to burn frequency as they suggest that water colouration/ DOC is related to the extent of recent burning (described under sub-question d) which is determined by rotation length; the shorter the rotation the greater the extent of recent burning. Thus, there is strong evidence (from six catchment studies and one model, see evidence under sub-question d) that increased frequency of burning results in an increase in water colouration/DOC.

8.16 No evidence was identified that specifically separated different effects related to variation in the size or location of burns on upland peatlands, although there is some evidence that larger fires tend to be more variable in terms of intensity and severity (for example, Tucker 2003 [4+]). They also reduce diversity at a fine scale in terms vegetation composition and structure compared to a mosaic of different-aged burns (for example, Legg & Davies 2009 [4+]), although this relates particularly to *Calluna* and heaths.

Research recommendations on the effects of differing characteristics of burns

- More consistent and systematic recording of information about the type and ideally intensity and severity of burns in research and monitoring projects. Related to this, consideration of the incorporation of such information about, or actual differing intensity/severity of burn treatments, including 'cool' and other burns, in experimental studies.

9 Effects of the interaction of managed burning and grazing on upland peatland biodiversity, carbon and water

9.1 The full text for sub-topic (f) is:

How does the interaction of managed burning and grazing affect upland peatland biodiversity carbon and water?

Introduction and context

9.2 Most moorland in England, and hence most upland peatland, especially the more extensive blanket bog, wet heath and valley mire habitats, is grazed. Nevertheless, blanket bog is often regarded as a climax or plagioclimax habitat that does not necessarily need to be grazed by livestock (for example, Rodwell *et al.* 1991). Although in England most of it is grazed, low stocking rates are generally recommended for conservation management (Rawes & Hobbs 1979, Coulson *et al.* 1992, SWT 1995, Shaw *et al.* 1996, O'Brien *et al.* 2007, Backshall *et al.* 2001, Glaves 2008). Some blanket bog and other peatland habitats have stock excluded completely, or in winter, for example as part of restoration management included under AE agreements (Natural England 2009, Nisbet & Glaves 2010) or for stock management reasons, for example, in some valley mires where stock, particularly cattle, are sometimes fenced out.

9.3 Tucker (2003) noted that "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 most critical interaction with grazing is in the immediate post-burn phase when heavy grazing may have an impact on the composition and structure of the post-burn succession. Livestock may also have an effect due to trampling as well as grazing, the impact of which will vary by stock type and perhaps breed (see Birnie & Hulme (1990) for a discussion on the effect of sheep hooves on blanket bog). *Sphagnum,* for example, is intolerant of repeated trampling (Slater & Agnew 1977), so grazing may affect post-burn recovery. The more general impacts of grazing on moorlands including upland peatlands are reviewed in the impact of moorland grazing and stocking rates topic of the Uplands Evidence Review (Martin *et al.* 2013).

The studies and evidence for the interaction of burning and grazing

9.4 Nine studies all from the UK provide evidence for sub-question (h): two from Scotland (Rhum and Skye) and the remainder from England where five were from Moor House NNR in the North Pennines, with another from Northumberland and one with sites in both the Peak District and Yorkshire Dales (both in the Pennines).

9.5 These studies are all described in more detail under the effects of burning on biodiversity, carbon and water under sub-questions a-d (see Sections 4-7) with the descriptions concentrating on the findings relating to the interaction between burning and grazing. All are experimental studies in which grazing was a treatment in a nested design also involving burning treatments. Many of the other studies of burning described under sub-questions a-d were on sites that were grazed (under a normal moorland management regime), so cover the

effect of burning in a grazed situation, but did not specifically consider the interaction with burning.

9.6 The main experimental study was again the Hard Hill burning and grazing experiment at Moor House NNR, with the results from the main vegetation studies (1961-2001), *Rubus chamaemorus* studies (1969), hydrology and water chemistry studies (2005-2008) and carbon dynamics (2003-2004) all considered. The applicability of these studies is considered under sub-question a (see paragraphs 4.13-4.17).

9.7 The detailed summaries of the individual studies and three reviews that relate to the sub-question are given in Appendix 9.

Applicability of the evidence for the interaction of burning and grazing to UK upland peatlands

9.8 The following section reviews the applicability of the evidence from the nine primary studies reviewed to the effects of the interaction of managed burning and grazing to UK and, in particular, English upland peatlands. It draws on the assessment of the external validity carried out in assessing the individual studies (see paragraphs 2.14-2.15) and the studies as a whole, including such factors as geographical location and how representative the habitat/vegetation type and intervention was of the study area(s) and nationally.

9.9 Nine studies all from the UK provide evidence for sub-question (h): two from Scotland (Rhum, Miles 1971; and Skye, Currall 1981) and the remainder from England where five were from Moor House NNR in the North Pennines (Hard Hill burning experiment vegetation and hydrology and water chemistry/quality studies, Taylor & Marks 1971/Marks & Taylor 1972, Ward *et al.* 2007, and Rawes & Williams 1973/Rawes & Hobbs 1979), with another in Northumberland (Ross *et al.* 2003) and one with sites in the Peak District and Yorkshire Dales (both in the Pennines; Marrs *et al.* 2004). Thus, the evidence is limited in terms of geographical distribution in relation to the extent of the upland peatland resource in England as a whole, ie all from the Pennines and Northumberland and all concerning single sites apart from one with two sites rather than covering larger geographical areas.

9.10 The Hard Hill experiment is long-term, with the initial burn of all treatment plots in 1954 and vegetation monitoring between 1961 and 2001 to date and now supplemented with recent carbon and water studies. The grazing treatments were unfenced, being open to normal grazing of the moor, with a fenced ungrazed control. Most of the other studies reviewed related to short-term experiments where grazing was one of, or the only, post-burn treatment. Most of these were related to management/control of *Molinia* dominance in small plots. Only two of the studies had controlled grazing treatments at two different stocking rates, with the others apparently open to 'normal' moorland grazing. In most of the studies grazing was at low or moderate stocking rates, whereas the effects of grazing are greater at high stocking rates (for example, Shaw *et al.* 1996, Nisbet & Glaves 2010, Martin *et al.* 2013), for example where stock are attracted to the post-burn flush of new growth. Indeed, burning is carried out by graziers to enable more even grazing across moorland and to enable grazing of otherwise impenetrable, dense dwarf-shrub vegetation.

9.11 All but one of the sites was sheep grazed, with the other grazed by red deer. Whilst sheep grazing is the norm across most English moorland, some upland peatlands are grazed by other livestock, particularly cattle and sometimes ponies, especially on the SW moors. In the main experiment at Hard Hill plots, sheep grazing was light (c.0.1 sheep/ha) and summer only, whereas all year round sheep grazing (though reduced in winter), typically at higher rates (for example, typically 1 ewe/ha (in summer including followers with a 25% reduction in winter) in 'Tier 1' maintenance ESA agreements and 0.67 ewe/ha in 'Tier 2' restoration agreements, Nisbet & Glaves 2010) and a minimum summer stocking rate of 0.5 ewe/ha in

summer in Uplands ELS, Natural England 2010) is more common on moorland except in restoration schemes/options.

Summary and discussion of the evidence on the interaction of burning and grazing

9.12 The following section presents a summary and brief discussion of the findings of the review of the evidence for the effects of the interaction of burning and grazing drawing primarily on the primary studies and systematic review presented in Appendix 9. Where there is clear evidence of specific effects of burning this is summarised as an evidence statement, given in bold, including its strength (see paragraphs 2.23-2.24) and the number of studies that contribute to it including their type and quality categorisation. The statements are ordered from broad to more specific effects. Some of the reviews summarised are also referred to in discussion of the statements.

9.13 Tucker (2003) [4+] noted that the "aspects of interactions (between burning and grazing)... have been little researched and are not understood in detail", a situation that appears to still be true today.

9.14 The relatively small number of evaluated studies that included grazing treatments produced few significant interactions between burning and grazing, though there are many studies that demonstrate significant effects of these two major moorland management practices separately. It is however possible that interactions may occur at a relatively large, for example, moorland grazing unit, scale and are not easy to pick up in smaller plots. For example, new growth, particularly of graminoids, after burning generally attracts stock. Thus, burning is specifically used for stock management to provide more even grazing. The extent, including size and distribution of burn patches as well as total area burnt can influence the level of grazing by stock and hence the impact on recovery for a given stocking rate (for example, Phillips 2012 [4+]).

9.15 The long-term experimental study (at Hard Hill, Moor House 1961-2001 [1++]) showed little evidence of any long-term overall effect of sheep grazing (Lee *et al.* 2013), albeit at a very low overall stocking rate and in a general area with a relatively tall *Calluna* canopy which sheep tend to avoid. However, there is moderate evidence of some interactions with sheep grazing in the initial period following burning, in particular a greater increase in the extent of bare ground (after burning) and an increase in grazing on, and reduction in cover of, *Rubus chamaemorus* compared to ungrazed treatments. In another grazing study elsewhere at Moor House (Rawes & Williams 1973/Rawes & Hobbs 1979 [2+]), heavy sheep grazing resulted in a similar loss of *Calluna* and rapid increase in *Eriophorum vaginatum* irrespective of whether burning occurred or not.

9.16 There is moderate evidence (Miles 1971, Ross *et al.* 2003 and Marrs *et al.* 2004, all [1+]) that burning results in increased grazing of *Molinia* by sheep and deer, but that this may be short-lived. Miles (1971 [1+]) showed that the effect of increased deer grazing on blanket bog/*Molinia* grassland was only apparent in the year following burning. On wet heath, Ross *et al.* (2003 [1+]) showed differences between two sheep stocking rates, with both resulting in the replacement of *Calluna* by graminoids, but heavy grazing favouring *Carex nigra* and light grazing *Molinia* and *Deschampsia flexuosa*. A review (Anderson *et al.* 2006 [4+]), concluded that "evidence from experimental studies demonstrated that burning without adequate grazing or over-frequent burning with or without grazing leads to what can become overwhelming *Molinia* dominance."

9.17 There is weak evidence (from one study: Currall 1981 [2+]) that burning on short rotations and/or heavy grazing after burning can lead to maintenance of the dense graminoid phase in wet heath (rather than its replacement by *Calluna*), but that high grazing intensity and low burning frequency pushes the balance in favour of *Juncus squarrosus* and *Nardus stricta*.

9.18 These studies generally support at least in part Tucker's (2003, [4+]) conclusion that "the effects of grazing (in combination with burning) 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." He further suggested that in general grazing and burning impacts appear to affect upland vegetation in similar ways. For example, increased burning frequencies and grazing intensities on heathland both tend to change vegetation from *Calluna* dominated communities to grasslands. 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. This is equally if not more true on upland peatlands.

Research recommendations on the interaction of burning and grazing

- The incorporation of grazing treatments in experimental burning studies, ideally at a range of grazing intensities including at rates typical of moorland grazing units incorporating upland peatland.

10 The relationship between managed burning of upland peatlands and 'wildfire'

10.1 The full text for sub-topic question (g) is:

Is there a relationship between managed burning of upland peatlands and 'wildfire' (risk, hazard, occurrence, severity, extent and damage)?

Introduction and context

10.2 The term 'wildfire' comes from north America where it refers to "unplanned, unwanted wildland fire including unauthorized human-caused fires, escaped wildland fire use events, escaped prescribed fire projects, and all other wildland fires where the objective is to put the fire out" (NWCG 2012). It has become relatively widely adopted in the UK where it tends to be used for out of control vegetation fires of whatever origin: 'accidental' fires resulting from (escaped) managed burns getting out of control; vandalism or neglect where fires are started deliberately (arson) or by neglect (for example, throwing away cigarettes, matches, broken glass or picnic fires); and 'natural' fires started by lightning or sunlight magnified by glass, although there is little evidence to indicate that this is a major cause in England at present (Bruce 2002).

10.3 Wildfires are of increasing concern in the UK as their frequency and magnitude are likely to increase with climate change (Solomon *et al.* 2007, Albertson 2010). McMorrow *et al.* (2009) in a comprehensive review of the wildfire problem in UK moorlands and heathlands, noted that "they pose a serious threat to the delivery of ecosystem goods and services, such as carbon retention, erosion prevention, or of provision of habitat for biodiversity, as well as threats to human settlements in some situations." Severe wildfires are not uncommon in the UK especially in drought years, particularly on moorland and heathland. However, they are not well recorded at a national scale, although some data are collated by the Fire Service and better data are available on their frequency and extent for some areas, notably the Peak District National Park (Anderson 1986, McMorrow *et al.* 2009). Wildfires are much more frequent in years of severe spring and summer drought, for example in spring 2003 when 152,600 fires were recorded (Asken 2004). Relatively little information appears to be collated on the characteristics of wildfires such as the cause, habitats/vegetation types involved, their extent, duration and severity, and frequency and return periods.

10.4 Wildfires tend to be larger and more severe than managed fires, though like managed fires they vary in severity. Nevertheless, they pose a threat to moorland ecosystem services, particularly of peatlands. Once ignited, peatland fires can burn for days, sometimes underground, and expose peat, initiate erosion, resulting in carbon loss, and be costly to restore. Severe wildfires can also damage peatland vegetation (Tallis 1973, 1987, Anderson 1987, Maltby *et al.* 1990, Gilchrist *et al.* 2004).

10.5 However, when managed burning is carried out there is always some risk, although little information appears to be available on the frequency of such accidental wildfires. Conversely, managed burning can be beneficial in reducing the fuel load of moorland vegetation, particularly of *Calluna*, and hence potentially reducing the risk of wildfire (Bruce 2002, McMorrow *et al.* 2009) and future wildfire hazard (Davies *et al.* 2006). Other potential approaches to reduce hazard as part of wildfire risk management, such as cutting or the restoration of modified, particularly *Calluna*-dominated, peatlands to active, functioning blanket mire vegetation, for example, through rewetting are beyond the scope of this topic

review. Tucker (2003) whilst recommending that blanket bogs and other mires should not be burnt, suggested a possible exception where "there is specific and clear overall conservation benefit, for example, in high fire risk areas, from a significant reduction in the risk of severe wildfires that may ignite peat layer."

The studies and evidence on the relationship between burning and wildfire

10.6 No studies were found that specifically provided evidence on the direct relationship between managed burning and occurrence and severity of 'wildfire' in the UK. However, relevant aspects of two studies and two reviews related to wildfire risk and hazard are presented in Appendix 10. Wider studies on the occurrence and characteristics, including severity and effects, of wildfires are beyond the scope of this review on managed burning effects, although a few wildfire studies are included under other sub-questions where they provided important information relevant to fire effects that was otherwise lacking.

10.7 Some of the evidence from studies reported under sub-question (e) on fire behaviour in relation to moisture content and fuel load are also relevant to fire risk that are relevant to wildfire and contributed to the development of evidence statements on this sub-question.

Applicability of the evidence on the relationship between burning and wildfire to UK upland peatlands

10.8 Whilst the increasing importance and interest in wildfire as a current and potentially increasing issue in view of climate change, very little evidence was found relating directly to the relationship between managed burning and wildfire. The two related studies and two reviews reviewed are all from the UK.

Summary and discussion on the relationship between burning and wildfire

10.9 The following section presents a summary and brief discussion of the findings of the review of the evidence on the relationship between managed burning and the occurrence and severity wildfire primarily from the studies and reviews described in Appendix 10 and under sub-question (e). Where there is clear evidence of specific effects of burning this is summarised as an evidence statement, given in bold, including its strength (see paragraphs 2.23-2.24) and the number of studies that contribute to it and their type and quality categorisation. The statements are ordered from broad to more specific effects. Some of the reviews summarised above are also referred to in discussion of the statements.

10.10 Although little evidence was found of a direct relationship between managed burning and the occurrence, severity and extent of wildfire, there is moderate evidence (Alberstson *et al.* 2009, 2010 [2+]; and the following reviewed under sub-question e: Davies (2005), Davies *et al.* 2010a and Davies & Legg 2011 [all 2+]; and three reviews/studies: Aylen [4-]; McMorrow *et al.* 2009 [2+]; Davies *et al.* 2008 [4+]; and Legg & Davies 2009 [4+] reviewed under sub-question e) that fuel load and structure are critical factors in fire behaviour and that burning can be, and is, used to reduce fuel load and hence fire hazard. Burning reduces fuel load and may therefore have benefits for fire risk management, alongside other measures such as cutting and/or the creation of a network of firebreaks and control zones.

10.11 Related to this, there is moderate evidence (Alberstson *et al.* 2009, 2010 [2+], Aylen [4-]; McMorrow *et al.* 2009 [2+]; and Davies *et al.* 2008 [4+]) that there may be an increased need for fire risk management in future, if certain climate change scenarios become a reality. This is likely to continue to include fuel management by burning or cutting and/or the creation of a network of fire breaks and fire control zones.

10.12 There is moderate evidence (McMorrow *et al.* 2009 [2+]) that 'heather moorland' in the Peak District which is mostly managed by rotational burning is less prone to the occurrence of wildfires than other moorland habitats. Although they still occur relatively frequently, this reflects the large extent of heather moorland and lower than predicted occurrence in relation to area.

10.13 Although outside the narrow scope of this sub-question in relation to wildfire risk, hazard, occurrence, severity, and extent, it is noted that Allen *et al.* (2013 [2+], reviewed under sub-question c) showed a clear interaction between managed burning rotation interval and wildfire return interval in terms of above-ground carbon balance. At 50 and 100 year wildfire return intervals, above-ground carbon losses were minimised by short prescribed burning rotations. However, under a 200 year wildfire return interval, above-ground carbon loss was minimised by long rotation intervals where delayed regeneration was modelled. However, in peatland habitats such as the study site, the amount of carbon in the peat greatly exceeds that in above ground vegetation (Gray & Levy 2009 [4+]).

Research recommendations on the relationship between burning and wildfire

- National collation of data on the occurrence and characteristics of wildfires, including the relationship with habitat and management including managed burning.
- Further studies on of the occurrence wildfire in relation to managed burning including specifically on upland peatland habitats, perhaps by extending the modelling work done in the Peak District.
- Review of evidence more generally on wildfire including frequency and extent, risk, severity and damage and management.

11 The extent, frequency, practice and type of managed burning on upland peatlands

11.1 The full text for sub-topic question (h) is:

What are the extent, frequency, practice and type of managed burning (including 'cool burning') on upland peatlands (including in relation to designated sites and water catchments)?

Introduction and context

11.2 Little systematic data is collected nationally on the extent and distribution of burning generally including on moorland in the uplands. What is available does not separate peatlands out from other moorland habitats with remote sensing studies often mapping burning on *Calluna*-dominated moorland as 'heather moorland' irrespective of the underlying heath or bog habitats.

11.3 More locally, for example in some National Parks (NP) and on some individual sites, mapping of moorland burning is more systematic. But given that different approaches and protocols are used which make collation difficult. Furthermore, as nationally, local mapping often does not separate out peatlands from other burned habitats. For these reasons, such mapping has generally not been included in this review.

11.4 Information on the frequency of burning is often determined from the same maps of extent if they are repeated at intervals or the age of burns can be determined from aerial photographs. Much less information still is available on the practice and types of managed burning in the English uplands and more widely. Some information can be collected on the severity of recent burns from field surveys. Habitat condition data based on Common Standards Monitoring (JNCC 2009) falls into this category. It includes some attributes directly related to recent burning and is collected by Natural England at intervals from all SSSIs and a proportion of AE agreements, but until recently not collated for individual attributes. Nevertheless, national summaries are periodically published on the condition of SSSIs in England including attributes relating to burning.

The studies and evidence on the extent, frequency and practice of managed burning

11.5 Nine studies all from the England provide evidence on sub-question (h). Most are national sample or census surveys, though others cover individual moorland areas: the Peak District NP/North Peak ESA, the North York Moors NP and the North Pennines AONB. The detailed individual summaries can be found in Appendix 11.

Applicability of the evidence on the extent, frequency and practice of managed burning to UK upland peatlands

11.6 National sample surveys of the English blanket bog resource using API provide a representative assessment of the extent and to a more limited degree the location of burning nationally for the time periods the photographs relate to. These probably include other upland peatland habitats, particularly wet heath, to an unknown extent, but there is no comparable

information available specifically for these other peatland habitats. Similarly, national sample field surveys of habitat condition provide a representative assessment of the frequency and to some extent severity of burning impacts and burning within 'sensitive areas'.

Summary and discussion of the evidence on the extent, frequency and practice of managed burning

11.7 The following section presents a summary of the findings of the review of the evidence on the extent, frequency and practice of managed burning primarily from the primary studies presented in Appendix 11. Where there is clear evidence of specific effects of burning this is summarised as an evidence statement, given in bold, including its strength (see paragraphs 2.23-2.24) and the number of studies that contribute to it and their type and quality categorisation. The statements are ordered from broad to more specific effects.

Extent

11.8 There is strong evidence (three [2++]: ADAS 1997 [2++], Penny Anderson Associates 2012 and Yallop *et al.* 2012; one [2+]: Yallop *et al.* 2006a; and one [2-]: Anderson *et al.* 2009) that the extent of rotational burning management is variable across upland peatlands in England and that it is mostly concentrated in the Pennines, Bowland and Northumberland. A recent national mapping exercise (Penny Anderson Associates 2012 [2++]) shows that it occurs on about a quarter of the total moorland deep peat resource in England. It is generally characterised by strip burning typically associated with grouse moors. Over the remainder of the resource burning on deep peat is now infrequent or does not normally take place, although it did occur on some of this land in the past (for example, Miller *et al.* 1984, Ward 1972, Ward *et al.* 1972, Defra in press a).

11.9 There is moderate evidence (one from an upland area, ADAS 1997 [2++]; and one a national sample survey: Yallop *et al.* 2006a [2+]) that there has been an increase in the extent (and frequency) of managed burning on moorland in England, including specifically on degraded 'dry' blanket bog in the North Peak ESA.

11.10 This is supported indirectly by moderate evidence (Natural England 2009 [2+]) of a recent increase in the number of gamekeepers employed (29%) and potential number of shooting days per year (29%) on grouse moors in the north of England, although this relates to all *Calluna*-dominated moorland rather than specifically peatlands.

Frequency

11.11 There is moderate evidence (one involving five datasets including a national sample survey: Yallop *et al.* 2012 [2++]; and others relating to individual upland areas: ADAS 1997 [2++] and Yallop *et al.* 2006a [2+]) that, there is considerable variation in the frequency of burning on upland peatlands. Nationally, the average return period was 64 year (or 26.5 year for the *Calluna*-dominated proportion). However, this includes the majority of upland peatland that is subject to little or no rotational burning. In areas where rotational burning occurs on deep peat, the proportion burnt per year and hence the average return period is higher than the national average: 21.2 year (11.7 year) in the North York Moors, 39.3 year (15 year) in the North Pennines and 73.1 year (25 year) in the Peak District (and 25% in another earlier study of the North Peak ESA).

11.12 There is moderate evidence that in most areas the proportion burnt per year on upland peatland and dry heath are similar and that the proportion burnt per year in the national dataset was similar on SSSIs and on (the smaller area of) non-designated upland peatland.

11.13 No evidence was identified on the coincidence of burning and water catchments.

Practice and type of burning

11.14 There is little evidence on the types of burning practice taking place in the English uplands in general and specifically on deep peat including on the extent to which 'cool burning' is practiced. There is, however, moderate evidence (from two national sample surveys: Critchley *et al.* 2011a, b, both [2++]) that burns into the bryophyte and lichen layer occur in a proportion of cases on blanket bog (11% of all, including unburned, samples) and wet heath (17%). These data suggest that that the severity of burning is variable, both within and between sites and that, in practice, burning into the bryophyte and lichen layer occurs in at least parts of burns in a proportion of cases.

Research recommendations on the extent, frequency and practice of managed burning

- Repeat remote sensing surveys to map changes in the extent and frequency of burning on upland peatlands, particularly blanket bog, nationally and in the main areas where burning occurs in the north of England.
- Collation and analysis of data from Natural England condition/integrated monitoring surveys particularly in relation to the burning related attributes.
- Improved recording of burning in condition assessments/integrated site assessments on upland peatland habitats.
- Repeat of detailed condition surveys of Priority Habitat Inventories, perhaps on a rolling programme with a proportion of new sites added.

12 Conclusions

12.1 In this topic review, evidence was identified in relation to eight sub-questions and summaries of evidence and evidence statements produced separately for each. The following sections summarise the evidence by sub-questions, and themes within them, by listing in simplified, slightly reordered form the evidence statements developed in Sections 4-11 with in some cases a brief explanation and/or interpretation.

12.2 A total of 54 evidence statements were developed from the evidence derived from 170 references grouped in to 123 evaluated studies. Of these, the majority were classed as strong, 19 (35%), or moderate, 28 (52%), with only six (11%) classed as weak and one (2%) as inconsistent. There are clear differences in the volume and strength of evidence across the eight sub-questions (Table 7), with a greater volume and stronger evidence on the effects on flora, fauna, carbon and water, and on burning extent, but less on the effects of differences in severity, interactions with grazing and wildfire. Recommendations for future research and other evidence gathering to address gaps are made at the end of this section.

Table 7 Number and strength of evidence statements by sub-question

Sub-question	Strong	Moderate	Weak	Inconsistent	Total
flora	6	5	1	0	12
fauna	5	2	2	0	9
carbon	2	4	0	0	6
water	2	4	2	1	9
severity	2	3	0	0	5
grazing	0	2	1	0	3
wildfire	0	3	0	0	3
extent	2	5	0	0	7
Total	**19**	**28**	**6**	**1**	**54**

The effects of burning on vegetation composition, structure and function

12.3 There is strong evidence that managed, rotational burning results, at least for a period, in a change in the species composition of blanket bog and upland wet heath vegetation. This included strong evidence:

- that burning of blanket bog and wet heath typically leads to an initial period of graminoid dominance, in particular of hare's-tail cottongrass *Eriophorum vaginatum*, purple moor-grass *Molinia caerulea* or deergrass *Trichophorum cespitosum*, typically lasting 10-20 years, and at least an initial decline in dwarf-shrub cover and in some cases diversity;
- that heather and some other dwarf shrubs tend to decline during the initial graminoid-dominant phase, but typically then increase, especially on drier sites, and may become dominant. This may take 15-20 year or longer on less-modified, wetter blanket bog and may not occur, for example, with too frequent or severe burning and/or heavy grazing;
- that bryophytes as a group tend to decline initially after burning of blanket bog. *Sphagnum* bog-mosses as a group have shown mixed responses, in some cases increasing in the

early post-burn stages, sometimes declining or being killed and sometimes then increasing or recolonising after varying periods; and
- that burning is associated with the creation of bare ground at least at a fine-scale.

12.4 There is moderate evidence:

- that burning leads to an increase in cloudberry *Rubus chamaemorus*;
- that the composition of blanket bog vegetation can continue to show change more than 80 year after the last burn; and
- that burns can lead the creation of relatively flat, unpatterned bog surfaces. This may be followed by the re-establishment of hummock-hollow topography following gradual recovery or recolonisation of *Sphagnum* bog-mosses.

12.5 There is comparatively little evidence on the effects of differences in burning rotations on peatlands, with only one long-term experimental study (at Moor House National Nature Reserve in the North Pennines) that has covered multiple rotations of differing lengths (10 and 20-year). This provides moderate evidence that differences in frequency of burning affect the vegetation composition and structure of blanket bog. At this site, more frequent burning has promoted dominance of hare's-tail cottongrass, with heather achieving higher cover under the longer rotation.

12.6 Changes in vegetation composition and structure may affect the functioning of the peatland ecosystem and hence have effects on associated ecosystem services which are reviewed in subsequent sub-questions. When interpreted in relation to the characteristic floristic composition, structure and function of upland peatland habitats, overall these vegetation responses to burning, in particular the tendency to dominance of graminoids and/or heather at different post-burn stages and depending on site conditions, may reduce the chance of maintaining active, functioning peatland. Similarly, where restoration is an objective for modified, degraded upland peatland habitats, burning may perpetuate dominance of graminoids or heather.

The effects on the maintenance and enhancement of the characteristic fauna of upland peatlands

Invertebrates

12.7 There is strong evidence that burning indirectly influences the invertebrate community composition of upland peatland habitats, typically benefiting open-ground species such as ground beetles and surface-active spiders. Many of the studies indicate an increase in overall species-richness or diversity and suggest that this occurs through increases in structural diversity at a relatively fine scale resulting from patchwork burning, and the presence of open/short swards amongst taller unburnt vegetation. Thus, conditions are provided for open ground species and for species that favour taller vegetation such as some web-spinning spiders. Many of the studies were carried out on modified upland peatlands and hence not all of the invertebrate species and assemblages are necessarily peatland species associated with less modified, functioning, high quality upland peatland habitats.

12.8 There is moderate evidence, that too frequent burning is likely to render peatland sites less suitable or unsuitable for the large heath butterfly *Coenonympha tullia* (a UK BAP species), but that occasional burning may be beneficial perhaps in favouring the larval foodplant, hare's-tail cottongrass *Eriophorum vaginatum*, and in reversing succession on at least some drier sites.

12.9 There is moderate evidence (reviewed under sub-question d) that burning is correlated with changes in the diversity and composition of aquatic invertebrate assemblages in

watercourses draining upland peatland catchments. These changes reflect declines in certain groups, especially mayflies and stoneflies, and increases in flies.

Breeding birds

12.10 There is strong evidence of correlations between moorland habitat types, their vegetation composition and structure, and densities of some moorland breeding birds, particularly waders. In relatively few studies has this been related directly to peatlands rather than moorland in general or specifically to burning practice (rather than burning and predator control together). This includes strong evidence:

- that certain species are associated with particular moorland vegetation characteristics: red grouse and stonechat with increasing heather cover; snipe and curlew with heterogeneity in vegetation structure; golden plover and skylark with short vegetation; waders with wet conditions; whinchat with dense vegetation; stonechat with tall vegetation; and meadow pipit with grass-heather mixes;
- of correlations between burning and/or predator control intensity and densities of some moorland breeding birds. Higher densities of red grouse, golden plover and curlew with increased burning/predator control were each shown in two studies; and higher densities of lapwing, redshank and ring ouzel each in single studies. Two studies showed lower densities of meadow pipit and single studies showed lower densities of skylark, wheatear and twite with increasing intensity of burning/predator control; and
- of overlap in the first half of April between the dates of first egg-laying of some moorland bird species and the legal burning season (which closes on 15 April in the uplands in England). In all but one case, it was the minority of first nest attempts (with nine species having more than 10% of first egg-laying attempts before 15 April). This indicates a potential vulnerability for those species that nest on the ground or in vegetation that is likely to be burnt (six species) rather than evidence of actual losses. There is also moderate evidence of earlier nesting over time for eight species, which may increase the proportion of first nest attempts by mid-April in future.

12.11 Only one study looked in detail at changes in numbers of breeding birds (for five waders) in relation to burning. This showed moderate evidence of greater declines in golden plover under more intensive (rather than less intensive) burning management and that curlew and lapwing declined more on '*Calluna*-dominated' plots than on 'bog' plots.

12.12 One study showed moderate evidence of an increase in breeding success and numbers of lapwing, golden plover, curlew and red grouse and breeding success of meadow pipit in response to legal predator control, indicating that such control contributes to the increases shown by some species on grouse moors in other studies probably in addition to any burning effects.

12.13 There is weak evidence of a correlation between burning and/or predator control intensity and overall diversity of moorland breeding birds, although the same study showed no relationship with species richness.

The effects of burning on carbon sequestration

12.14 There is strong evidence that managed burning affects various components of the carbon cycle of upland peatlands. This includes strong evidence:

- that moorland burning results in increased water colouration and/or dissolved organic carbon (DOC) n in peatland watercourses (reviwed under sub-question c); and

moderate evidence:

- that burning reduces peat accumulation and reduces above and below ground carbon storage compared to no burning;
- that managed burning can result in erosion and reduction in the level of the soil surface;
- of increases in gross CO_2 fluxes of respiration and photosynthesis; and
- of carbon loses through fuel consumption during burning and in conversion to char.

12.15 Only relatively recently have attempts been made to estimate complete carbon budgets that consider the overall impacts of burning, some using modelling approaches. So far, these have produced inconsistent evidence, with predictions of both positive and negative overall effects of burning, although they provide strong evidence that burning affects the processes controlling carbon budgets of upland peatlands.

The effects of burning on water quality and flow

Water colouration/DOC

12.16 There is strong evidence that moorland burning results in increased water colouration and/or dissolved organic carbon (DOC) in peatland watercourses (which is also relevant to sub-question c). Related to this there is:

- strong evidence that the area of recent burning on deep peat is correlated with an increase in water colouration and/or DOC at the catchment-scale in watercourses draining peatland catchments;
- moderate evidence that the area of heather-dominated vegetation on deep peat is correlated with an increase in water colouration and/or DOC, in soil water in one case and in watercourses draining peatland catchments in another; and
- moderate evidence from laboratory studies that burning is associated with an increase in water colouration and increased pH (which are likely to be related as pH controls solubility of DOC).

12.17 However, the relatively small number of recent small plot- or stand-scale studies of water colouration and/or DOC in relation to burning have shown inconsistent evidence. It has been suggested that this may reflect differences in time since burning (as effects have been shown to be greatest soon after burning) and sampling too deep in the peat (as effects have been shown to occur only in the upper layer).

Soil and water chemistry

12.18 In relation to soil and water chemistry, there is weak evidence of differences in concentrations of chemical species after a burn, for example, with aluminium, iron and sodium increasing and calcium, chlorine and bromine, and pH declining.

12.19 As noted under sub-question (b), but also relevant to water quality, there is moderate evidence that burning is correlated with changes in the diversity and composition of aquatic invertebrate assemblages in watercourses draining upland peatland catchments (also relevant to sub-question b). These changes reflect declines in certain groups, especially mayflies and stoneflies, and increases in flies.

Watercourse aquatic invertebrates

12.20 As noted under sub-question (b), but also relevant to water quality, there is moderate evidence that burning is correlated with changes in the diversity and composition of aquatic invertebrate assemblages in watercourses draining upland peatland catchments (also relevant to sub-question b). These changes reflect declines in certain groups, especially mayflies and stoneflies, and increases in flies.

Hydrology and water flow

12.21 There is weak evidence from small-scale plot/stand studies: of shallower water tables initially after burning; and of increased frequency of surface runoff after recent burning. However, no evidence was identified specifically relating to the effect of burning on watercourse flow or the risk of downstream flood events. If there are any effects, they are likely to be highly site specific.

The effects of differences in severity, frequency, scale, location and other characteristics of burns

12.22 Few studies were identified that related differences in the severity and other characteristics of burns directly to differences in effects on the four aspects of upland peatlands reviewed under the main sub-questions (a-d). However, there is:

- strong evidence that moisture content, vegetation type and phenology, recent weather and human factors are important factors in the ignition of fires; and
- moderate evidence that fuel load and structure are critical factors in fire behaviour, particularly in 'fireline' intensity (heat output per unit length of fire front) and rate of spread, although residence time and depth of penetration of lethal temperatures into the soil are perhaps more important in determining severity of impact, but are much less well understood.

12.23 Little evidence was identified on the types of burning practice taking place in the English uplands in general and specifically on deep peat, including on the extent to which 'cool burning' is practiced. However, there is moderate, recent evidence (reviewed under sub-question h) that burns into the bryophyte and lichen layer are not infrequent on blanket bog and wet heath.

12.24 There is evidence that, in addition to initial fire severity, pre-fire vegetation composition is an important factor influencing post-fire recovery. Related to this, there is weak evidence (reviewed under sub-question a) that differences in the frequency of burning (ie rotation or return period) affect vegetation composition and structure on blanket bog. More frequent burning tends to promote dominance of single competitive species, particularly graminoids, especially hare's-trail cottongrass and purple moor-grass, and in drier situations, heather.

12.25 Some of the catchment-scale studies (reviewed under sub-question d) are relevant to burn frequency as they suggest that water colouration and/or DOC are related to the area of recent burning which is determined by rotation length; the shorter the rotation, the greater the proportion that has recently been burnt. Thus, there is strong evidence that increased frequency of burning results in an increase in water colouration and/or DOC.

12.26 No evidence was found that specifically identified differences in effects related to differences in the size or location of burns on upland peatlands, although more generally there is some evidence that larger fires tend to be more variable in terms of intensity and severity. They also reduce diversity at a fine scale in terms vegetation composition and structure compared to a mosaic of different-aged burns, although this relates particularly to heather and heaths.

The interaction between burning and grazing

12.27 The relatively small number of evaluated studies that included grazing treatments produced few s significant interactions between burning and grazing, although there are many studies that demonstrate significant effects of these two major moorland management practices separately (see also the UER topic report on the impacts of grazing and stocking rates, Martin *et al.* 2013). It is however possible that interactions may occur at a relatively large, for example, moorland grazing unit, scale and are not easy to pick up in smaller plots. For

example, new growth, particularly of graminoids, following burning generally attracts stock. Thus, burning is specifically used for stock management to provide more even grazing. The extent, including the size and distribution of burn patches as well as total area burnt, can influence the distribution and level of grazing by stock. However, there is:

- moderate evidence from the only long-term burning and grazing experiment (at Moor House) of some interactions with grazing in the initial period following burning: in particular a greater increase in the extent of bare ground and an increase in grazing on, and reduction in cover of, cloudberry, compared to ungrazed treatments;
- moderate evidence that burning results in increased grazing of purple moor-grass by sheep and deer, but that this may be short-lived; and
- weak evidence that burning on short rotations and/or heavy grazing after burning can lead to maintenance of the dense graminoid phase in wet heath (rather than its replacement by heather), but that high grazing intensity and low burning frequency pushes the balance in favour of heath rush and mat-grass.

Is there a relationship between managed burning and wildfire?

12.28 Although little evidence was found of a direct relationship between managed burning and the occurrence, severity and extent of wildfire, there is moderate evidence that fuel load and structure are critical factors in fire behaviour. Burning reduces fuel load and may therefore have benefits for fire risk management, alongside other measures such as cutting and/or the creation of a network of firebreaks and control zones. There may be an increased need for fire risk management in future, under some climate change scenarios. There is moderate evidence that 'heather moorland' in the Peak District which is mostly managed by rotational burning is less prone to the occurrence of wildfires than other moorland habitats. Although they still occur relatively frequently, this reflects the large extent of heather moorland and lower than predicted occurrence in relation to area.

The extent, frequency and practice of managed burning

Extent

12.29 There is strong evidence from a recent national mapping exercise that rotational management burning occurs on about a quarter of the total moorland deep peat resource in England, mostly in the Pennines, Bowland and Northumberland. Over the remainder of the resource, burning on deep peat is now infrequent or does not normally take place, although it did occur on some of this land in the past.

12.30 There is moderate evidence that there has been an increase in the extent (and frequency) of managed burning on moorland in England, including specifically on degraded 'dry' blanket bog in the North Peak ESA. There is also moderate evidence of a recent increase in the number of gamekeeper's employed and potential number of shooting days per year (both 29%) on grouse moors in the north of England, although this relates to all heather-dominated moorland rather than specifically peatlands.

Frequency

12.31 There is moderate evidence that, in areas where rotational burning occurs on deep peat, the proportion burnt per year and hence rotation length is relatively high and in some cases little different to or even higher than on dry heath in the same areas.

12.32 There is moderate evidence that, there is considerable variation in the frequency of burning on upland peatlands:

- Nationally, the average burn 'return period' (for heather-dominated proportion in brackets) was 64 year (26.5 year). However, this includes the majority of upland peatland that is subject to little or no rotational burning.
- In areas where rotational burning occurs on deep peat, the proportion burnt per year and hence the average return period is higher: 21.2 year (11.7 year) in the North York Moors, 39.3 year (15 year) in the North Pennines and 73.1 year (25 year) in the Peak District (and 25% in another earlier study of heather-dominated 'dry bog' in the North Peak ESA).
- In most of these areas the proportion burnt per year on upland peatland and dry heath are similar
- The proportion burnt per year in the national dataset was similar on SSSIs and on (the smaller area of) non-designated upland peatland.

12.33 No evidence was identified on the coincidence of burning and water catchments.

Practice and type of burning

12.34 There is little evidence on the types of burning practice taking place in the English uplands in general and specifically on deep peat, including on the extent to which 'cool burning' is practiced. There is, however, moderate evidence that burns into the bryophyte and lichen layer and in to sensitive areas occur in a proportion of cases on blanket bog and wet heath (in 11-17% of all, including unburned, samples in two national surveys).

Research recommendations

12.35 Overall, the evidence available on the effects of managed burning on upland peatlands is incomplete. Through this assessment of the available evidence, the following subjects and associated research recommendations are made:

12.36 Assessment of the available evidence indicates that the following areas would benefit from further research:

- The extension of experimental and other monitoring studies of the effects of burning on vegetation and ecosystem services to a wider range of sites across the English upland peatland resource, ideally including additional medium/long-term studies covering multiple rotations across the full length of typical blanket bog burn rotations (eg15-25 year) (which are currently restricted to the Hard Hill experiment at Moor House). Ideally these should consider type of burning (for example, 'cool' and other burns). Such studies should also include the wider range of upland peatland habitats including wet heath, flushes, fens (including valley mires) and swamps, and consider the interaction of burning and grazing across the range of typical stocking rates and regimes that occur in moorland grazing units that include peatland habitats. This has been done in part through the nearly completed NERC-funded EMBER (Effects of Moorland Burning on the Ecohydrology of River basins) project[6].
- Research on post-burn recovery times in upland peatlands including palaeo-archival studies on vegetation recovery after fire. Research on the effects of burning on the range of characteristic upland peatland species, especially individual *Sphagnum* bog-moss species, including post-burn recovery.
- Improved more detailed and consistent description of the characteristics of study sites, for example, in terms of habitat, degree of modification, vegetation composition (including *Sphagnum* species) and structure, surface topography and condition, not just in vegetation but in wider studies, for example, on carbon and water. In addition, also

[6] See: **www.geog.leeds.ac.uk/research/rbpm/projects/ember-effects-of-moorland-burning-on-the-ecohydrology-of-river-basins/**

recording information about the type and ideally intensity and/or severity of burns in related research projects.

- Improved and more consistent interpretation of existing and new vegetation data from an ecological and nature conservation/biodiversity perspective, for example, including consideration of aspects of autecology, functional types and associations, disturbance, habitats and vegetation community types, habitat condition, associated species, structure (including micro topography) and function.
- Research on restoration management, including the potential use of one-off burning and alternative treatments to reduce graminoid and heather dominance where this is an objective. This is being addressed at two sites in the Pennines and one at Bowland, including the effects on carbon and water, in a current Defra Environmental Stewardship research project, BD5104[7].
- Research on the effects of burning on key characteristic blanket bog species of fauna particularly invertebrates, reptiles and birds (including food availability, for example, craneflies as an important food item for waders).
- Further examination of data on bird nesting dates and breeding success in relation to burning (for example, from Nest Record Cards, vulnerability/risk from burning (especially short-eared owl and stonechat) and pre-nesting activity timing).
- Further studies addressing the relative lack of information on gaseous exchange of peatlands in relation to burning and on char production during burning and its significance.
- Extension of studies on aquatic invertebrates more widely across the English uplands. Interpretation of changes in community composition in terms of water quality and biodiversity, possibly including as food availability for predators (for example, fish and birds such as dipper).
- Studies of the effects of differences in the intensity/severity of fires and characteristics of burn patches such size, shape, location (for example, in relation to slope, watercourses etc), distribution etc.
- National collation of data on the occurrence and characteristics of wildfires, including the relationship with managed burning and further study of the occurrence of wildfire in relation to managed burning on upland peatlands, perhaps by extending the modelling work done in the Peak District.
- Repeat of remote sensing surveys to map changes in the extent and frequency of burning on upland peatlands, particularly blanket bog, nationally and in the main areas where burning occurs in the north of England.
- Definitive, agreed mapping of grouse moors, together with data on burning management, for correlation studies, particularly with breeding bird survey data, and the relationship to other land uses including water catchments and designated sites.
- Improved recording of the occurrence and severity/effects of burning and wildfires in site surveys of upland peatland habitats, for example in Natural England's condition assessment/'integrated site assessment'. National collation and analysis of data from Natural England's condition/integrated monitoring surveys particularly in relation to burning-related attributes. A repeat of the national sample survey of more detailed condition assessment of upland habitats in the Priority Habitat Inventories (last done in 2008-10), perhaps on a rolling programme with a proportion of new sites added to the existing sites.

[7] See: **https://sites.google.com/a/york.ac.uk/peatlandesuk/**

13 Glossary of terms

The following list of technical terms used in the report draws on a range of sources including Davies *et al*. (2008) and FAO (1986).

Term	
API	Aerial photographic interpretation, for example, of habitats or burns.
BAP	Biodiversity Action Plan.
Bog	Ombrogenous mire.
Bulk density	Also known as dry bulk density. The mass of dry material, per unit volume.
Catchment	The area upslope of a point, line or area, towards which all surface water drains (for example, the catchment of the grip) OR an area where all the surface water drains towards a common point. Often the same thing.
CH_4	Methane.
CO_2	Carbon dioxide.
Conductivity (1)	Hydrological conductivity = a measure of the inherent properties of a material that control how quickly water will move through them.
Conductivity (2)	Electrical conductivity, used in testing solutions (soil water, streams etc) to indicate the concentration of a range of solutes, interacting with other chemical properties.
DOC	Dissolved Organic Carbon.
Fen	Mire receiving water from sources other than precipitation.
Fire danger	An assessment of both fixed and variable factors of the environment that determine the ease of ignition, rate of spread,, difficulty of control and fire impact.
Fire hazard	Measure of that part of the fire danger contributed by the fuels available for burning, determined by the relative amount, type and condition, particularly moisture content.
Fire regime	The pattern of occurrence, size and severity (and sometimes also vegetation and fire effects) in a given area or ecosystem.
Fire risk	The probability of fire initiation due to the presence and activity of a causative agent.
Fire severity	The degree to which a site has been altered or disrupted by fire.
Fireline intensity (or intensity or fire intensity)	The rate of heat release per unit time per unit length of fire front. The product of heat from combustion, quantity of fuel consumed per unit area of fire front and the rate of spread of a fire, expressed in kW m^{-1}.
Flashiness	The extent to which a flow of water is flashy.

Table continued…

Term	
Flashy (of hydrographs during rainfall events)	Responding quickly by increases in flow to the onset in the catchment of rainfall, maximum rain deposition, and by decreases in flow to cessation or reduction in rainfall intensity.
GHG	Greenhouse gas (CO_2, N_2O, CH_4).
GPR	Ground Penetrating Radar.
Groundwater	Water held in the bedrock, drift and soils forming a continuous mass in one or all of these.
Gully	A channel caused by erosion of a peat mass, which may be branched or linear, and may be found entirely within the peat mass, or cutting through into underlying mineral material (also gullying, gullied).
Hagg	A remnant block of undisturbed peat that has been separated from the rest of the peat mass by anastomosing gullies.
Hydrograph	A record showing the flow rate (volume/time) of a stream or channel at a given point, over time.
Macrofossil	Literally large fossils, used in peat stratigraphy, however, to denote recognisable plant remains, usually requiring microscopy.
Mire	Habitat which forms peat.
Moorland Line	Definition of semi-natural moorland vegetation in the uplands (Less Favoured Areas) produced for MAFF (now Defra) by aerial photographic interpretation (API) with 'ground truthing'.
N_2O	Nitrous oxide.
NEE	Net Ecosystem Exchange (of CO_2).
Ombrogenous	Formed due to the influence of precipitation.
Ombrotrophic (of a habitat or ecosystem)	Receiving all its nutrient supply from precipitation or atmostpheric deposition.
PAR	Photosynthetically active radiation.
Peat	(i) the partially decomposed remains of plants and other organisms which have accumulated in waterlogged conditions, at the surface of the soil profile or as material infilling water bodies. (ii) a soil texture class encompassing any soil material with greater than 20-30% organic matter (depending on clay content).
pH	A measure of the acidity or alkalinity of a solution or material.
POC	Particulate Organic Carbon.
SCP	Spheroidal Carbonaceous Particles – soot particles found in peat deposits associated with industrial activity.
Sphagnum	A genus of mosses characterised by whorled branched growth form, also called bog-mosses.

Table continued…

Term	
Water table	The distance of a waterlogged zone beneath the surface. Note that in many bogs this water table is not the ground water, in that it is discontinuous with a more freely drained zoned at its base. However, in raised bogs, water table in bogs may be continuous with ground water, but largely not interacting, because the water table is raised bog the gravitational level of the ground water due to low hydrological conductivity of peat.
White moor	Moorland with *Calluna* largely absent and usually dominated by grasses, particularly *Molinia*.
Wildfire	Any unplanned and uncontrolled wildland fire that, regardless of ignition source, may require suppression response, or other action according to agency policy.

14 References

Evaluated studies included in the report

ADAMSON, J. & KAHL, J. 2003. *Changes in vegetation at Moor House within sheep exclosure plots established between 1953 and 1972*. Merlewood: CEH report to English Nature.

ADAS. 1997. *Biological monitoring of moorland in the North Peak ESA 1988–1996*. London: ADAS.

ALBERTSON, K., AYLEN, J., CAVAN, G. & MCMORROW, J. 2009. Forecasting the outbreak of moorland wildfires in the English Peak District. *Journal of Environmental Management,* 90, 2642-2651.

ALBERTSON, K., AYLEN, J., CAVAN, G. & MCMORROW, J. 2010. Climate change and the future occurrence of moorland wildfires in the Peak District of the UK. *Climate Research,* 45, 105-118.

ALLEN, S.E. 1964. Chemical aspects of heather burning. *Journal of Applied Ecology,* 1, 347-367.

ALLEN, K.A., HARRIS, M.P.K. & MARRS, R.H. 2013 Matrix modelling of prescribed burning in *Calluna vulgaris*-dominated moorland: short burning rotationsminimize carbon loss at increased wildfire frequencies. *Journal of Applied Ecology*, 2013, doi: 10.1111/1365-2664.12075.

AMAR, A., GRANT, M., BUCHANAN, G., SIM, I., WILSON, J., PEARCE-HIGGINS, J.W. & REDPATH, S. 2011. Exploring the relationships between wader declines and current land-use in the British uplands. *Bird Study,* 58, 13-26.

ANDERSON, B.J., ARROYO, B.E.,COLLINGHAM, Y.C., ETHERIDGE, B., FERNANDEZ-DE-SIMON, J., GILLINGS, S., GREGORY, R.D., LECKIE, F.M., SIM, I.M.W., THOMAS, C.D., TRAVIS, J.M. J. & REDPATH, S.M. 2009. Using distribution models to test alternative hypotheses about a species' environmental limits and recovery prospects. *Biological Conservation*, 142, 488-499.

ANDERSON, P., ROSS, S., EYRE, G. & LONGDEN, K. 2006. Restoration of *Molinia*-dominated blanket mires. *CCW Report*, 742. Bangor: CCW.

ARMSTRONG, A., HOLDEN, J. & STEVENS, C. 2009. *The differential response of vegetation to grip-blocking*. Leeds: University of Leeds report to North Pennines AONB..

ARMSTRONG, A., HOLDEN, J., LUXTON, K. & QUINTON, J.N. 2012. Multi-scale relationship between peatland vegetation type and dissolved organic carbon concentration. *Ecological Engineering*, 47, 182-188.

ASPRAY, K.L. 2012. *Macroinvertebrate communities and ecosystem functioning in peatland streams*. Leeds: PhD thesis, University of Leeds.

AYLEN, J., CAVAN, G. & ALBERTSON, K. 2007. *Identifying the best strategy for mitigating moorland wildfire risk*. Manchester: University of Manchester report to Moors for the Future.

BEHARRY-BORG, N., HUBACEK, K., TERMANSEN, M., SMART, J., CHAPMAN, P.J., ROBROEK, B.J.M., HOLDEN, J., IRVINE, B.J., KIRKBY, M.J., ASHLEY, D., CHAPMAN, D.S., QUINN, C.H., REED, M.S. & KAY, P. 2009. *Determining the socio-economic implications of different land management strategies in Yorkshire Water's catchments*. Leeds: University of Leeds report for Yorkshire Water.

BENSCOTER, B. W., THOMPSON, D.K., WADDINGTON, J.M., FLANNIGAN, M.D., WOTTON, B.M., DE GROOT, W.J. & TURETSKY, M.R. 2011. Interactive effects of vegetation, soil moisture and bulk density on depth of burning of thick organic soils. *International Journal of Wildland Fire,* 20, 418-429.

BROWN, A. & BAINBRIDGE, I.P. 1990. Grouse moors and upland breeding birds. In: Thompson, D.B.A., Hester, A.J. & Usher, M.B. (eds.) *Heaths and moorland: cultural landscapes*, pp 51-66. Edinburgh: HMSO.

BROWN, L., HOLDEN, J., RAMCHUNDER, S. & LANGTON, R. 2009. Understanding moorland aquatic insect ecology to inform biodiversity conservation and sustainable land management. Leeds: University of Leeds report to the North Pennines AONB Peatscapes Partnership.

BURCH, J. 2008. The relationship of bryophyte regeneration to heather canopy height following moorland burning on the North York Moors. *Journal of Bryology,* 30, 208-216.

BURCH, J. 2009. The regeneration of bryophytes after the burning of dry heath and wet heath moorland on the North York Moors. *Natural England Comissioned Report*, NECR011*.* Sheffield: Natural England.

CHAMBERS, F.M., MAUQUOY, D., GENT, A., PEARSON, F., DANIELL, J. R.G. & JONES, P.S. 2007. Palaeoecology of degraded blanket mire in South Wales: Data to inform conservation management. *Biological Conservation,* 137, 197-209.

CHAPMAN, D.S., TERMANSEN, M., QUINN, C.H., JIN, N. L., BONN, A., CORNELL, S.J., FRASER, E.D.G., HUBACEK, K., KUNIN, W.E. & REED, M.S. 2009. Modelling the coupled dynamics of moorland management and upland vegetation. *Journal of Applied Ecology,* 46, 278-288.

CHAPMAN, P.J., MCDONALD, A.T., TYSON, R., PALMER, S.M., MITCHELL, G. & IRVINE, B. 2010b. Changes in water colour between 1986 and 2006 in the headwaters of the River Nidd, Yorkshire, UK. *Biogeochemistry,* 101.

CHAPMAN, P.J., PALMER, S.M., IRVINE, B.J., MITCHELL, G. & MCDONALD, A.T. 2011. A response to 'Changes in water colour between 1986 and 2006 in the headwaters of the River Nidd, Yorkshire, UK: a critique of methodological approaches and measurement of burning management' by Yallop *et al*. *Biogeochemistry*.

CHEN, Y., MCNAMARA, N.P., DUMONT, M.G., BODROSSY, L., STRALIS-PAUESE, N. & MURRELL, J.C. 2008. The impact of burning and *Calluna* removal on below-ground methanotroph diversity and activity in a peatland soil. *Applied Soil Ecology,* 40, 291-298.CLAY, G. D. 2009. *The Impacts of Heather and Grassland Burning inthe Uplands: Creating Sustainable Strategies.* Durham: PhD, University of Durham.

CLAY, G.D. & WORRALL, F. 2011. Charcoal production in a UK moorland wildfire - How important is it? *Journal of Environmental Management*, 92, 676-682.

CLAY, G.D., WORRALL, F. & AEBISCHER, N.J. 2012. Does prescribed burning on peat soils influence DOC concentrations in soil and runoff waters? Results from a 10 year chronosequence. *Journal of Hydrology,* 448, 139-148.

CLAY, G.D., WORRALL, F., CLARK, E. & FRASER, E.D.G. 2009a. Hydrological responses to managed burning and grazing in an upland blanket bog. *Journal of Hydrology,* 376, 486-495.

CLAY, G.D., WORRALL, F. & FRASER, E.D.G. 2009b. Effects of managed burning upon dissolved organic carbon (DOC) in soil water and runoff water following a managed burn of a UK blanket bog. *Journal of Hydrology,* 367, 41-51.

CLAY, G.D., WORRALL, F. & FRASER, E.D.G. 2010a. Compositional changes in soil water and runoff water following managed burning on a UK upland blanket bog. *Journal of Hydrology,* 380, 135-145.

CLAY, G.D., WORRALL, F. & ROSE, R. 2010b. Carbon budgets of an upland blanket bog managed by prescribed fire. *Journal of Geophysical Research-Biogeosciences,* 115, 14.

CLUTTERBUCK, B. 2009. *Land management (specifically controlled heather burning) as a factor controlling carbon loss from upland peat soils.* Cranfield. PhD thesis, Cranfield University.

CLUTTERBUCK, B. & YALLOP, A. R. 2010. Land management as a factor controlling dissolved organic carbon release from upland peat soils. 2: Changes in DOC productivity over four decades. *Science of the Total Environment,* 408, 6179-6191.

COTTON, D.E. & HALE, W.H.G. 1994. Effectiveness of cutting as an alternative to burning in the management of *Calluna vulgaris* moorland: results of an experimental field trial. *Journal of Environmental Management,* 40, 155-159.

COULSON, J.C. 1988. The structure and importance of invertebrate communities on peatlands and moorlands, and effects of environmental and management changes. *Special Publication of the British Ecological Society,* 7.?

COULSON, J.C. & BUTTERFIELD, J. 1986. The spider communities on peat and upland grasslands in northern England. *Holarctic Ecology*, 9, 229-239.

COULSON, J.C., FIELDING, C.A. & GOODYEAR, S.A. 1992. The management of moorland areas to enhance their nature conservation interest. *Joint Nature Conservation Committee Reports.* Peterborough: Joint Nature Conservation Committee.

COUWENBERG, J., THIELE, A., TANNEBERGER, F., AUGUSTIN, J., BÄRISCH, S., DUBOVIK, D., LIASHCHYNSKAYA, N., MICHAELIS, D., MINKE, M., SKURATOVICH, A. & JOOSTEN, H. 2011. Assessing greenhouse gas emissions from peatlands using vegetation as a proxy. *Hydrobiologia,* 674, 67-89.

CRICK, H., MOSS, D. JOYS, A.& CLARK, J. 2006. Moorland burning and birds. *BTO News*, 263, 4-5.

CRITCHLEY, N. (ed.). 2011a. *Condition surveys of upland priority habitats in England: blanket bog*. Sheffield: ADAS report to Natural England.

CRITCHLEY, N. (ed.). 2011b. *Condition surveys of upland priority habitats in England: wet heath*. Sheffield: ADAS report to Natural England.

CURRALL, J.E.P. 1981. *Some effects of management by fire on wet heath vegetation in western Scotland.* Aberdeen: PhD thesis, University of Aberdeen.

CURRALL, J.E.P. 1989. *Some* effects of management by fire on wet heath vegetation in western Scotland*. Some effects of management by fire on wet heath vegetation in western Scotland*. *Dissertation Abstracts International*, 49, 11, May 1989.

CURTIS, D.J. & CORRIGAN, H. 1990. Peatland spider communities and land management on a Scottish island. *Bulletin de la Societe Europeenne d'Arachnologie Hors Serie*, 97-102.

DAPLYN, J. & EWALD, J. 2006. *Birds, burning and grouse moor management*. Hope Valley: Report to Moors for the Future.

DAVIES, G.M. 2001. *The impact of muirburning on lichen diversity.* Edinburgh: MSc thesis, University of Edinburgh.

DAVIES, G.M. 2005. *Fire behaviour and impact on heather moorland.* Edinburgh: PhD thesis, University of Edinburgh.

DAVIES, G.M., GRAY, A., HAMILTON, A. & LEGG, C.J. 2008. The future of fire management in the British uplands. *International Journal of Biodiversity Science & Management,* 4, 127-147.

DAVIES, G.M. & LEGG, C. J. 2008. The effect of traditional management burning on lichen diversity. *Applied Vegetation Science,* 11, 529-538.

DAVIES, G.M. & LEGG, C.J. 2011. Fuel moisture thresholds in the flammability of *Calluna vulgaris*. *Fire Technology,* 47, 421-436.

DAVIES, G.M., LEGG, C.J., O'HARA, R., MACDONALD, A.J. & SMITH, A.A. 2010. Winter desiccation and rapid changes in the live fuel moisture content of *Calluna vulgaris*. *Plant Ecology & Diversity,* 3, 289-299.

DAVIES, G. M., LEGG, C. J., SMITH, A. A. & MACDONALD, A. J. 2009. Rate of spread of fires in *Calluna vulgaris*-dominated moorlands. *Journal of Applied Ecology,* 46, 1054-1063.

DENNIS, R.L. H. & EALES, H.T. 1997. Patch occupancy in *Coenonympha tullia* (Muller, 1764) (Lepidoptera: Satyrinae): habitat quality matters as much as patch size and isolation. *Journal of Insect Conservation,* 1, 167-176.

DENNIS, R. L.H. & EALES, H.T. 1999. Probability of site occupancy in the large heath butterfly *Coenonympha tullia* determined from geographical and ecological data. *Biological Conservation,* 87, 295-301.

ELLIOTT, R.J. 1953. *The effects of burning on the heather moors of the South Pennines.* Sheffield: PhD thesis, University of Sheffield.

ELLIS, C.J. 2008. Interactions between hydrology, burning and contrasting plant groups during the millennial-scale development of sub-montane wet heath. *Journal of Vegetation Science,* 19, 693-U57.

EYRE, M.D., LUFF, M.L. & WOODWARD, J.C. 2003. Grouse moor management: habitat and conservation implications for invertebrates in southern Scotland. *Journal of Insect Conservation,* 7, 21-32.

FARAGE, P., BALL, A., MCGENITY, T.J., WHITBY, C. & PRETTY, J. 2009. Burning management and carbon sequestration of upland heather moorland in the UK. *Australian Journal of Soil Research,* 47, 351-361.

FARAGE, P., BALL, A.S., MCGENITY, T.J., WHITBY, C. & PRETTY, J. 2010. Reply to Comment on: 'Burning management and carbon sequestration of upland heather moorland in the UK'. *Australian Journal of Soil Research,* 48, 104-104.

FLETCHER, K., AEBISCHER, N.J., BAINES, D., FOSTER, R. & HOODLESS, A.N. 2010. Changes in breeding success and abundance of ground-nesting moorland birds in relation to the experimental deployment of legal predator control. *Journal of Applied Ecology*, 47, 263-272.

FORREST, G.I. 1971. Structure and production of north Pennine blanket bog vegetation. *Journal of Ecology,* 58, 13-49.

FORREST, G.I. & SMITH, R.A.H. 1975. The productivity of a range of blanket bog vegetation types in the Northern Pennines. *Journal of Ecology,* 63, 173-199.

FULLEN, M.A. 1983. Some changes in air temperature and wind velocity after burning of heather (*Calluna vulgaris*) moor and their relation to moorland surface processes. *Naturalist,* 108, 17-24.

GARNETT, M.H., INESON, P. & STEVENSON, A.C. 2000. Effects of burning and grazing on carbon sequestration in a Pennine blanket bog, UK. *Holocene,* 10, 729-736.

GARNETT, M.H., INESON, P., STEVENSON, A.C. & HOWARD, D.C. 2001. Terrestrial organic carbon storage in a British moorland. *Global Change Biology,* 7, 375-388.

GLAVES, D.J., HAYCOCK, N.E., COSTIGAN, P., COULSON, J.C., MARRS, R.H., ROBERTSON, P.A. & YOUNGER, J. 2005. *Defra review of the Heather and Grass Burning Regulations and Code: Science Panel assessment of the effects of burning on biodiversity, soils and hydrology*. London: Defra.

GRAND-CLEMENT, E. 2008. *Heather burning in peatland environments: effects on soil organic matter and peat accumulation.* Reading: PhD thesis, University of Reading.

GRANT, M.C., MALLORD, J., LEIGH, S. & THOMPSON, P.S. 2012. The costs and benefits of grouse moor management to biodiversity and aspects of the wider environment: a review. *RSPB Research Report.* Sandy, Bedfordshire: RSPB.

GRAY, A. & LEVY, P. 2009. *A review of carbon flux research in UK peatlands in relation to fire and the Cairngorms National Park*. Grantown on Spey: CEH report for Cairngorms National Park.

GRAYSON, R., HOLDEN, J. & ROSE, R. 2010. Long-term change in storm hydrographs in response to peatland vegetation change. *Journal of Hydrology,* 389, 336-343.

GRAYSON, R., KAY, P., FOULGER, M. & GLEDHILL, S. 2012. A GIS based MCE model for identifying water colour generation potential in UK upland drinking water supply catchments. *Journal of Hydrology,* 420, 37-45.

GWCT. 2010. *Waders on the fringe: Why nationally scarce waders flourish on grouse moors*. Fordingbridge: Game & Wildlife Conservation Trust.

HALE, W. H.G. & COTTON, D.E. 1988. The management of vegetation change on Ilkley Moor. *Aspects of Applied Biology,* 16, 311-316.

HAMILTON, A. 2000. *The characteristics and effects of management fire on blanket-bog vegetation in north-west Scotland.* Edinburgh: PhD thesis, University of Edinburgh.

HARRIS, M.P.K. 2011. *The role of prescribed burning: moorland management in the Peak District.* Liverpool*:* PhD thesis, University of Liverpool.

HARRIS, M.P.K., ALLEN, K.A., LE DUC, M.G., EYRE, G. & MARRS, R.H. 2011a. Prescribed fire characteristics and biomass reduction on upland moorland. *Aspects of Applied Biology*, 171-177.

HARRIS, M.P.K., ALLEN, K.A., MCALLISTER, H.A., EYRE, G., LE DUC, M.G. & MARRS, R.H. 2011b. Factors affecting moorland plant communities and component species in relation to prescribed burning. *Journal of Applied Ecology,* 48, 1411-1421.

HARRIS, M.P.K., GEMMELL, C., MCALLISTER, H., LE DUC, M.G. & MARRS, R.H. 2006. *The effects of cool burning on the vegetation at Howden and Bamford Moor in the Peak District*. Liverpool: University of Liverpool report to Moors for the Future.

HAWORTH, P.F. & THOMPSON, D.B.A. 1990. Factors associated with the breeding distribution of upland birds in the south Pennines, England. *Journal of Applied Ecology,* 27, 562-577.

HOBBS, R.J. 1981. *Post-fire succession in heathland communities*. Aberdeen: PhD thesis, University of Aberdeen.

HOBBS, R.J. 1984. Length of burning rotation and community composition in high-level *Calluna-Eriophorum* bog in N England. *Vegetatio,* 57, 129-136.

HOBBS, R.J. & GIMINGHAM, C.H. 1980. Some effects of fire and grazing on heath vegetation. *Bulletin d'Ecologie,* 11, 709-715.

HOCHKIRCH, A. & ADORF, F., F. 2007. Effects of prescribed burning and wildfires on *Orthoptera* in Central European peat bogs. *Environmental Conservation,* 34, 225-235.

HOLDEN, J. 2005a. Controls of soil pipe frequency in upland blanket peat. *Journal of Geophysical Research-Earth Surface,* 110, 11.

HOLDEN, J. 2005b. Piping and woody plants in peatlands: cause or effect? *Water Resources Research,* 41, W06009.

HOLDEN, J., CHAPMAN, P. J., PALMER, S. M., KAY, P. & GRAYSON, R. 2011. *A review of moorland burning impacts on raw water quality with a focus on water colour*. Rep. to Yorkshire Water Services. Leeds: University of Leeds.

HOLDEN, J., CHAPMAN, P.J., PALMER, S.M., KAY, P. & GRAYSON, R. 2012. The impacts of prescribed moorland burning on water colour and dissolved organic carbon: A critical synthesis. *Journal of Environmental Management,* 101, 92-103.

HOLDEN, J., WEARING, C., PALMER, S., JOHNSTON, K. & BROWN, L. 2013. Fire decreases near-surface hydraulic conductivity and macropore flow in blanket peat. *Hydrological Processes*, 2013, doi: 10.1002/hyp.9875.

HOLMES, P.R., BOYCE, D.C. & REED, D.K. 1993. The ground beetle (Coleoptera, Carabidae) fauna of welsh peatland biotopes - factors influencing the distribution of ground beetles and conservation implications. *Biological Conservation,* 63, 153-161.

IUCN. 2011. *Burning and peatbogs*. Edinburgh: IUCN UK Peatland Programme.

JNCC. 2009. *Common Standards Monitoring guidance for upland habitats*. Version July 2009. Peterborough: Joint Nature Conservation Committee.

JNCC. 2011. Towards an assessment of the state of UK peatlands. *JNCC Report*, 445*.* Peterborough: JNCC.

JONES, P.S. 2005. Burning on blanket bog: a scientific perspective. In: Sherry, J. *Guidelines for burning and cutting dwarf-shrub vegetation on lowland and upland heath and blanket bog. Appendix 1*,10-18. Bangor: Countryside Council for Wales.

KINAKO, P.D. S. & GIMINGHAM, C.H. 1980. Heather burning and soil erosion on upland heaths in Scotland. *Journal of Environmental Management,* 10, 277-284.

KIRKPATRICK, J. B., MARSDEN-SMEDLEY, J. B. & LEONARD, S. W. J. 2011. Influence of grazing and vegetation type on post-fire flammability. *Journal of Applied Ecology,* 48.

LEE, H.,. ALDAY, J.G., ROSE, R.J., O'REILLY, J. & MARRS, R.H. 2013. Long-term effects of prescribed burning and low-intensity sheep grazing on blanket bog plant communities. *Journal of Applied Ecology*, 2013, doi: 10.1111/1365-2664.12078.

LEGG, C.J. & DAVIES, G.M. 2009. What determines fire occurrence, fire behaviour and fire effects in heathlands? In: Proceedings of the 10th National Heathland Conference - managing heathlands in the face of climate change. *Natural England Commissioned Report*, NECR014, pp 45-55. Sheffield: Natural England.

LINDSAY, R. 2010. *Peatbogs and carbon: a critical synthesis*. London: University of East London.

LINDSAY, R.A. 1977. Glasson Moss NNR and the 1976 fire: A report describing the effects of the 1976 fire on the vegetation of this Grade 1 site. *NCC CST Report* NCC.

LINDSAY, R.A. & ROSS, S.Y. Monitoring of peat bog systems. *Monitoring of Ecological Change in Wetlands of Middle Europe, 1994 Linz, Austria* pp 73-92. Slimbridge: IWRB Publication, 30.

LINDSAY, R., CHARMAN, D.J., EVERINGHAM, F., O'REILLY, R.M., PALMER, M.A., ROWELL, T.A. & STROUD, D.A. 1988. *The Flow Country: The peatlands of Caithness and Sutherland.* JNCC Technical Report. Peterborough: Joint Nature Conservation Committee.

LITTLEWOOD, N., ANDERSON, P., ARTZ, R., BRAGG, O., LUNT, P. & MARRS, R. 2011. *Peatland biodiversity*. Edinburgh: Report to IUCN UK Peatland Programme.

LOFTUS, M.C.P. 1994. *The ecology and management of upland vegetation in the Wicklow Mountains.* Belfast: D Phil thesis, New University of Ulster.

LUNT, P., ALLOTT, T., ANDERSON, T., BUCKLER, M., COUPAR, A., JONES, P., LABADTZ, J. & WORRAL, P. 2011. *Peatland restoration*. Edinburgh: Report to IUCN UK Peatland Programme.

MACDONALD, A.J. 2008. Fire and compaction as management tools on raised bogs. *Scottish Natural Heritage Commissioned Report.* Edinburgh: Scottish Natural Heritage.

MARKS, T.C. & TAYLOR, K. 1972. The mineral nutrient status of *Rubus chamaemorus* l. In relation to burning and sheep grazing. *Journal of Applied Ecology*, 9, 501-511.

MARRS, R.H., PHILLIPS, J.D.P., TODD, P.A., GHORBANI, J. & LE DUC, M.G. 2004. Control of *Molinia caerulea* on upland moors. *Journal of Applied Ecology*, 41, 398-411.

MCDONALD, A.T., NADEN, P.S., MITCHELL, G. & MARTIN, D. 1991. *Discoloured water investigations*. Leeds: University of Leeds Report to Yorkshire Water.

MCFERRAN, D. M., MCADAM, J.H. & MONTGOMERY, W.I. 1995. The impact of burning and grazing on heathland plants and invertebrates in Country Antrim. *Preceedings of the Royal Irish Academy,* 95B, 1-17.

MCMORROW, J., LINDLEY, S., AYLEN, S., CAVAN, J. G., ALBERTSON, K. & BOYS, D. 2008. Moorland wildfire risk, visitors and climate change: patterns, prevention and policy. *In:* BONN, A., ALLOTT, T., HUBACEK, K. & STEWART, J. (eds.) *Drivers of change in upland environments*, pp 404-431*.* London: Routledge.

MILES, J. 1971. Burning *Molinia*–dominant vegetation for grazing by red deer. *Grass and Forage Science*, 26, 247-250.

MILLER, C.J. 2008. *Mechanisms of water colour release from organic soilsand consequences for catchment management*. Aberdeen: PhD thesis, University of Aberdeen.

MITCHELL, G. & MCDONALD, A.T. 1995. Catchment characterisation as a tool for upland water-quality management. *Journal of Environmental Management,* 44, 83-95.

MOSS, D., JOYS, A.C., CLARK, J.A., KIRBY, A., SMITH, A., BAINES, D. & CRICK, H.Q.P. 2005. Timing of breeding of moorland birds. *BTO Research Report*, 362*.* Thetford.

MOWFORTH, M.A. & SYDES, C. 1989. Moorland management: a literature review. *Research and Survey in Nature Conservation*, 25*.* Peterborough: NCC.

NATURAL ENGLAND. 2009. *Mapping values; the vital nature of our uplands. An atlas linking environment and people*. Sheffield: Natural England.

NATURAL ENGLAND. 2010. *England's peatlands: carbon storage and greenhouse gases*. Sheffield: Natural England.

O'BRIEN, H., LABADZ, J. & BUTCHER, D. 2005. *An investigation of the impact of prescribed moorland burning in the Derwent catchment upon discolouration of surface waters*. Nottingham: Nottingham Trent University report for Moors for the Future.

O'BRIEN, H.E. 2009. *The role of blanket peat moorland management in the generation and amelioration of discolouration of surface water supplies*. Nottingham: PhD thesis, Nottingham Trent University.

ORWIN, K.H. & OSTLE, N.J. 2012. Moss species effects on peatland carbon cycling after fire. *Functional Ecology,* 26, 829-836.

O'REILLY, C. 2008. *Peatscapes project: Sphagna as management indicators research*. Lambley: Ptyxis Ecology report to North Pennines AONB Partnership.

PATTISON, I. & LANE, S. 2011. The link between land-use management and fluvial flood risk: A chaotic conception? *Progress in Physical Geography,* 36, 72–92.

PEARCE-HIGGINS, J.W. & GRANT, M.C. 2006. Relationships between bird abundance and the composition and structure of moorland vegetation. *Bird Study,* 53, 112-125.

PEARCE-HIGGINS, J. W., GRANT, M.C., BEALE, C.M., BUCHANAN, G.W. & SIM, I.M.W. 2009. International importance and drivers of change of upland bird populations. *In:* BONN, A., ALLOTT, T., HUBACEK, K. & STEWART, J. (eds.) *Drivers of environmental change in uplands* pp 209-227. London: Routledge.

PENNY ANDERSON ASSOCIATES. 2012. Mapping the status of upland peat using aerial photographs. *Natural England Commissioned Report*, NECR089. Sheffield: Natural England.

PICOZZI, N. 1968. Grouse bags in relation to the management and geology of heather moors. *Journal of Applied Ecology*, 5, 483-488.

PIETIKÄINEN, J., TURUNEN, J. & TOLONEN, K. 1999. The role of fire in the carbon dynamics of a mire, eastern Finland. *Holocene*, 9, 453-462.

RAMCHUNDER, S.J. 2010. *The effects of artificial drainage, drain-blocking and burning on peatland stream ecosystems* Leeds: PhD thesis, University of Leeds.

RAMCHUNDER, S.J., BROWN, L.E. & HOLDEN, J. 2009. Environmental effects of drainage, drain-blocking and prescribed vegetation burning in UK upland peatlands. *Progress in Physical Geography,* 33, 49-79.

RAMCHUNDER, S.J., Brown, L.E. & Holden., J. 2013. Rotational vegetation burning effects on peatland stream ecosystems. *Journal of Applied Ecology*, 2013, doi: 10.1111/1365-2664.12082.

RATCLIFFE, D.A. 1990. *Bird life of mountain and upland*. Cambridge: Cambridge University Press.

RAWES, M. 1983. Changes in two high altitude blanket bogs after the cessation of sheep grazing. *Journal of Ecology,* 71, 219-235.

RAWES, M. & HOBBS, R. 1979. Management of semi-natural blanket bog in the Northern Pennines. *Journal of Ecology,* 67, 789-807.

RAWES, M. & WILLIAMS, R. 1973. Production and utilisation of *Calluna* and *Eriophorum*. *Potassium Institute ltd Colloquium Proceedings*, 3, 115-119.

ROSS, S., ADAMSON, H. & MOON, A. 2003. Evaluating management techniques for controlling *Molinia caerulea* and enhancing *Calluna vulgaris* on upland wet heathland in Northern England, UK. *Agriculture Ecosystems & Environment,* 97, 39-49.

ROWELL, T.A. 1988. The Peatland management handbook. *Research and Survey in Nature Conservation*, 14*.* Peterborough: Nature Conservancy Council.

ROWELL, T.A. 1990. Management of peatlands for conservation. *British Wildlife,* 1, 144-156.

SHAW, S., WHEELER, B. & BACKSHALL, J. 1997. Review of effects of burning and grazing blanket bogs; conservation issues and conflicts. *In:* TALLIS, J. H., MEADE, R. & HULME, P. D. (eds.) *Blanket Mire Degradation - Causes, Consequences and Challenges.* Aberdeen: Macaulay Land Use Research Institute on behalf of the Mires Research Group.

SHAW, S.C., WHEELER, B.D., KIRBY, P., PHILIPSON, P. & EDMUNDS, R. 1996. Literature review of the historical effects of burning and grazing of blanket bog and upland wet heath. *English Nature Research Report,* 172. Peterbrough: English Nature.

SHETLER, G., TURETSKY, M. R., KANE, E. S. & KASISCHKE, E. 2008. *Sphagnum* mosses limit total carbon consumption during fire in Alaskan black spruce forests. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere*, 38, 2328-2336.

SMITH, A.A., REDPATH, S.M., CAMPBELL, S.T. & THIRGOOD, S.J. 2001. Meadow pipits, red grouse and the habitat characteristics of managed grouse moors. *Journal of Applied Ecology,* 38, 390-400.

SOTHERTON, N., MAY, R., EWALD, J., FLETCHER, K. & NEWBORNE, D. Managing uplands for game and sporting interests: an industry perspective. In: BONN, A., ALLOTT, B., HUBACEK, K. & STEWART, J. (eds.) *Drivers of Environmental change in uplands,* pp 241-260. London: Routledge

STEWART, G.B., COLES, C.F. & PULLIN, A.S. 2004. *Does burning degrade blanket bog? Systematic Review 1.* Birmingham: Centre for Evidence-Based Conservation, University of Birmingham.

STEWART, G.B., COLES, C.F. & PULLIN, A.S. 2005. Applying evidence-based practice in conservation management: Lessons from the first systematic review and dissemination projects. *Biological Conservation,* 126, 270-278.

STONE, M. 2006. *Impact of heather burning on vegetation and carabid communities in the Peak District National Park*. Leeds: MSc thesis, University of Leeds.

TAYLOR, K. & MARKS, T.C. 1971. The influence of burning on growth and development of *Rubus chamaemorus* L. *Calluna-Eriophorum* bog. In: Duffey, E. & Watt, A.S. (eds.) *The scientific management of animal and plant communities for conservation*, pp 153-166. Oxford: Blackwell Scientific.

THARME, A.P., GREEN, R.E., BAINES, D., BAINBRIDGE, I.P. & O'BRIEN, M. 2001. The effect of management for red grouse shooting on the population density of breeding birds on heather-dominated moorland. *Journal of Applied Ecology,* 38, 439-457.

THOMPSON, D.B.A., MACDONALD, A.J., MARSDEN, J.H. & GALBRAITH, C.A. 1995b. Upland heather moorland in Great Britain: a review of international importance, vegetation change and some objectives for nature conservation. *Biological Conservation,* 71, 163-178.

TUCKER, G. 2003. Review of the impacts on heather and grassland burning in the uplands on soils, hydrology and biodiversity. *English Nature Research Reports,* 550. Peterborough: English Nature.

TUCKER, G. 2004a. The burning of uplands and its effect on wildlife. *British Wildlife,* 15.

TUCKER, G. 2004b. The impacts of burning in upland habitats. In: Anderson, P (ed.) *Upland ecology, tourism and access*, pp 65-78. Proceedings of the 18th conference of the IEEM, Buxton, 25-27 November 2003. Winchester: Institute of Ecology and Environmental Management.

USHER, M. B. 1992. Management and diversity of arthropods in *Calluna* heathland. *Biodiversity and Conservation,* 1, 63-79.

WARD, S.E., BARDGETT, R.D., MCNAMARA, N.P., ADAMSON, J.K. & OSTLE, N.J. 2007. Long-term consequences of grazing and burning on northern peatland carbon dynamics. *Ecosystems,* 10, 1069-1083.

WHITE, S. 2004. *Control of water quality from upland catchments. Summary report*. Cranfield: Cranfield University report to Yorkshire Water. [Submitted to the review by Yorkshire Water.]

WORRALL, F. & ADAMSON, J. K. 2008. The effect of burning and sheep grazing on soil water composition in a blanket bog: evidence for soil structural changes? *Hydrological Processes,* 22, 2531-2541.

WORRALL, F., ARMSTRONG, A. & ADAMSON, J.K. 2007. The effects of burning and sheep-grazing on water table depth and soil water quality in a upland peat. *Journal of Hydrology*, 339, 1-14.

WORRALL, F., BELL, M.J. & BHOGAL, A. 2010a. Assessing the probability of carbon and greenhouse gas benefit from the management of peat soils. *Science of the Total Environment,* 408, 2657-2666.

WORRALL, F., CLAY, G.D., MARRS, R. & REED, M.S. 2011a. *Impacts of burning managament on peatlands*. Edinburgh: report to IUCN UK Peatland Programme.

WORRALL, F., ROWSON, J. & DIXON, S. 2012. Effects of managed burning in comparison to vegetation cutting on dissolved organic carbon concentrations in peat soils. *Hydrological Processes,* 2012, doi: 10.1002/hyp.9474.

WORRALL, F. & WARBURTON, J. 2009. *Assessing successful strategies for grip-blocking in the North Pennines AONB*. Report to North Pennines AONB.

YALLOP, A.R. & CLUTTERBUCK, B. 2009. Land management as a factor controlling dissolved organic carbon release from upland peat soils 1: spatial variation in DOC productivity. *Journal of Biology,* 7, 3803-3813.

YALLOP, A. R., CLUTTERBUCK, B. & THACKER, J. 2010. Increases in humic dissolved organic carbon export from upland peat catchments: the role of temperature, declining sulphur deposition and changes in land management. *Climate Research,* 45, 43-56.

YALLOP, A.R., THACKER, J. & CLUTTERBUCK, B. 2006a. Mapping extent of burn management in the North Pennines: Review of extent year 2001-2003. *English Nature Research Report*, 698*.* Peterborough: English Nature.

YALLOP, A.R., THACKER, J.I., THOMAS, G., STEPHENS, M., CLUTTERBUCK, B., BREWER, T. & SANNIER, C.A.D. 2006b. The extent and intensity of management burning in the English uplands. *Journal of Applied Ecology,* 43, 1138-1148.

YALLOP, A.R., THOMAS, G., THACKER, J., BREWER, T. & SANNIER, C. 2005. A history of burning as a management tool in the English uplands. 1: estimates of areal extent of management burning in English uplands. *English Nature Research Report*, 667. Peterborough: English Nature.

YALLOP, A.R., WHITE, S.M. & CLUTTERBUCK, B. 2008. Evidence for a mechanism driving recent observed trends in dissolved organic carbon release from upland peat soils. *Aspects of Applied Biology*, 85, 127-132.

YALLOP, A.R., THACKER, J. & CLUTTERBUCK, B. 2012. *Burning on deep peat and bog habitat in England: reconciliation and re-examination of results from English Nature Research Reports 667, 698 and unpublished data*. Cranfield: Cranfield University report to Natural England.

Additional references quoted in the report as context

These references were not formally evaluated and hence they did not contribute to the development of evidence statements.

ADAS. 1993. *Moorland map of England 1992*. London: ADAS report to MAFF.

ANDERSON, P. 1986. *Accidental moorland fires in the Peak District: a study of their incidence and ecological implications*. Bakewell: report for the Peak District Moorland Restoration Project, Peak Park Joint Planning Board.

ANDERSON, P. 1997. Fire damage on blanket mires. *In:* TALLIS, J. H., MEADE, R. & HULME, P. D. (eds.) *Blanket mire degradation: causes, consequences and challenges.* Aberdeen: Macaulay Land Use Research Institute on behalf of the Mires Research Group.

ASKEN LTD. 2004. *Fire outbreaks in spring 2003*. Kendal: Asken Ltd report to the Countryside Agency, Countryside Council for Wales and the Forestry Commission.

ASCOLI, D., MARZANO, R. & BOVIO, G. 2006. Experimental fires for heather moorland management in northwestern Italy. *Forest Ecology and Management*, 234S, S1.

ATHERTON, I., BOSANQUET, S. & LAWLEY, M. (eds.). 2010. *Mosses and liverworts of Britain and Ireland: a field guide*. British Bryological Society.

AVERIS, A., AVERIS, B., BIRKS, J., HORSFIELD, D., THOMPSON, D. & YEO, M. 2004. *An illustrated guide to British upland vegetation,* Peterborough: Joint Nature Conservation Committee.

BACKSHALL, J., MANLEY, J. & REBANE, M. 2001. *The upland management handbook,* Peterborough: English Nature.

BAILEY, M.P. 2003. Reserve focus - Cors Fochno (Borth Bog), West Wales. *British Wildlife*, 14, 195-198.

BAIN, C. G., BONN, A. & STONEMAN, R. 2011. IUCN UK *Commission of Inquiry on Peatlands*. Edinburgh: IUCN UK Peatland Programme.

BAKER, J., SUCKLING, J. & CAREY, R. 2004. *Status of the adder Vipera berus and slow-worm Anguis fragilis in England*. English Nature Research Report, 546). Peterborough: English Nature.

BATES, S. 1993. *Burning on blanket bog on Dartmoor*. Exeter: English Nature.

BEEBEE, T. & GRIFFITHS, R. 2000. *Amphibians and reptiles: a natural history of the British herpetofauna*. London: Collins New Naturalist.

BELL, J.N.B. & TALLIS, J.H. 1973. Biological flora of the British Isles: *Empetrum nigrum* L. *Journal of Ecology*, 61,289-305.

BIRNIE, R.V. & HULME, P.D. 1990. Overgrazing of peatland vegetation in Shetland. *Scottish Geographical Magazine*, 106, 28-36.

BOSTOCK, J.L. 1980. The *history of the vegetation of the Berwyn mountains, north Wales*. Manchester: PhD thesis, University of Manchester.

BRUCE, M.A. 2002. Country report from the United Kingdom. *Baltex Fire 2000*. The second Baltic seminar and excercise in forest fire and information and resources exchange 2000. URL: **www.fire.uni-freiburg.de/iffn/country/gb/gb_2.htm** [Accessed May 2013].

BYRAM G.M. 1959. Combustion of forest fuels. In: Davis K.P. (ed.). *Forest Fire Control and Use*, pp 61-123. London: McGraw-Hill.

CATCHPOLE, E.A. & CATCHPOLE, W.R. 1991. Modelling moisture damping for fire spread in a mixture of live and dead fuels. *International Journal of Wildland Fire*, 1, 101-106.

CCW. 1999. *Tir Gofal management prescriptions*. Bangor: Countryside Council for Wales.

CHAMBERS, F.M., MAUQUOY, D., PEARSON, F., DANIELL, J.R.G., GENT, A. & COOK, C. 2000. *Recent vegetational change in Welsh blanket mires: a palaeoecological appraisal*. CCW Contract Science Report No. 420. Bangor: Countryside Council for Wales.

CHARMAN, D. 2002. *Peatlands and environmental change*. Chichester: Wiley.

CHENEY, N.P., GOULD J.S. & CATCHPOLE W.R. 1993. The influence of fuel, weather and fire shape variables on fire-spread in grasslands. *International Journal of Wildland Fire*, 3, 31-44.

COLLINS, S.L. & ADAMS, D.E. 1983. Succession in grasslands: thirty-two years of change in a central Oklahoma tallgrass prarie. *Vegetatio*, 51, 181-190.

CONWAY,V.M. & MILLAR, A. 1960. The hydrology of some small peat-covered catchments in the northern Pennines. *Journal of the Institution of water Engineers*, 14, 415-424.

CORBETT, G.B. & HARRIS, S. 1991. *The handbook of British mammals*. Oxford: Blackwell.

CRICK, H.Q.P., DUDLEY, C., GLUE, D.E., & THOMSON, D.L. 1997. UK birds are laying earlier. *Nature*, 388, 526.

CRICK, H.Q.P. & SPARKS, T.H. 1999. Climate change related to egg-laying trends. *Nature*, 399,423-424.

CROWLE, A. 2007. Letting our carbon go free: the sustainable management of carbon and blanket peat in the English uplands. *British Wildlife*, 19, 28-34.

DARDNI. No date. *Heather Moorland*. Belfast: Department of Agriculture and Rural Development, Northern Ireland.

DAVIES, G.M., SMITH, A.A., MACDONALD, A.J., BAKKER, J.D. & LEGG, C.J. 2010b. Fire intensity, fire severity and ecosystem response in heathlands: factors affecting the regeneration of *Calluna vulgaris*. *Journal of Applied Ecology,* 47, 356-365.

DEFRA. 2007. *The Heather and Grass Burning code: 2007 version*. London: Defra.

DEFRA. 2008a. *The Heather and Grass Burning code 2007. Best practice guide 3: identifying sensitive areas*. London: Defra.

DEFRA. 2010. *Assembling UK wide data on soil carbon (and greenhouse gas) in the context of land management*. London: Defra.

DEFRA. 2011. *Project specification: restoration of blanket bog vegetation for biodiversity, carbon sequestration and water regulation*. London: Defra.

DEFRA. In press a. *The Heather and Grass Burning Code 2007. Best practice guide 7: Burning in the uplands of south-west England.* London: Defra.

DEFRA. In press b. *The Heather and Grass Burning Code 2007. Best practice guide 8: Identifying and managing heather beetle damage*. London: Defra.

EALES, H.T. & DENNIS, R.L.H. 1998. Predicting site occupancy for *Coenonympha tullia* (Müller, 1764) (Lepidoptera: Nymphalidae, Satyrinae) using habitat parameters. *Entomologist's Gazette*, 49, 3-16.

EDDY, A., WELCH, D. & RAWES, M. 1969. The vegetation of the Moor House National Nature Reserve in the northern Pennines, England. *Vegetatio,* 16, 239-284.

ENGLISH NATURE & MOORLAND ASSOCIATION. 2005. *Management plan template for grouse moor estates/SSSI*. Peterborough: English Nature & the Moorland Association.

EVANS, C.D., MONTEITH, D.T. & COOPER, D M. 2005. Long-term increases in surface water dissolved organic carbon: Observations, possible causes and environmental impacts. *Journal of Environmental Pollution*, 137, 55–71.

EVANS, F.C. & DAHL, E. 1955. The vegetational structure of an abandoned field in Southeastern Michigan and its relation to environmental factors. *Ecology*, 36, 685-706.

EVANS, D.M., REDPATH, S.M., EVANS, S.A., ELSTON, D.A., GARDNER, C.J. DENNIS, P. & PAKEMAN, R. 2006. Low intensity, mixed livestock grazing improves the breeding abundance of a common insectivorous passerine. *Biology Letters*, 2, 636-638.

FAO. 1986. Wildland fire management terminology. *FAO Forestry Paper*, 70. Rome: UN FAO.

FERNANDES, P.M. 2001. Fire spread prediction in shrub fuels in Portugal. *Forest Ecology and Management*, 144, 67-74.

FORGEARD, F. & FRENOT, Y. 1996. Effects of burning on heathland soil chemical properties: An experimental study on the effect of heating and ash deposits. *Journal of Applied Ecology,* 33.

FOX, R., ASHER, J. BRERETON, T., ROY, D. & WARREN, M. 2006. *The state of butterflies in Britain and Ireland*. Newbury: Pisces Publications for Butterfly Conservation.

FRAZER, D. 1983. *Reptiles and amphibians in Britain*. London: Collins New Naturalist.

FRASER, G.K. 1933. Scottish moorlands in relation to tree growth. *Forestry Commission Bulletin*, 15. HMS0.

FREEMAN, C., EVANS, C.D. & MONTEITH, D.T. 2001. Export of organic carbon from peatland soils. *Nature*, 414, 785.

FULLER, R.J., WARD, E., HIRD, D & BROWN, A.F. 2002. Declines of ground-nesting birds in two areas of upland farmland in the South Pennines of England. *Bird Study*, 49, 146-152.

GARDNER, S.M. & Usher, M.B. 1989. Insect abundance on burned and cut upland *Calluna* heath. *Entomologist*, 108, 147-157.

GILCHRIST, P., GILBERT, J. & BUTT, K. 2004. Burning issues: lessons from natural regeneration after wildfire. In: Anderson, P. (ed.). *Upland ecology, tourism and access*, pp 79-91. Proceedings of the 18th conference of the IEEM, Buxton, 25-27 November 2003. Winchester: IEEM.

GIMINGHAM, C.H. 1960.Biological flora of the British Isles: *Calluna*. Journal of Ecology, 48, 455-483.

GIMINGHAM, C H. 1971. *Calluna* heathlands: use and conservation in the light of soem ecological effects of management. In: Duffey, E. & Watt, A.S. (eds.) *The scientific management of animal and plant communities for conservation*, pp 91-103. Oxford: Blackwell Scientific.

GIMINGHAM, C H. 1972. *Ecology of heathlands,* London: Chapman and Hall.

GLAVES, D.J. 2008. *Review of guidance on sustainable moorland stocking rates particularly relating to blanket mire and other habitats with low carrying capacity*. Exeter: Natural England.

GRAY, A. 2006. *The influence ofmanagement on the vegetation and carbon fluxes of blanket bog.* Edinburgh: PhD, University of Edinburgh.

GRAYSON, R., HOLDEN, J. & ROSE, R. 2010. Long-term change in storm hydrographs in response to peatland vegetation change. *Journal of Hydrology,* 389, 336-343.

GRIME, J.P., HODGSON, J.G. & HUNT, R. 1988. *Comparative Plant Ecology.* London: Unwin Hyman.

HARTFORD, R.A. & FRANSDEN W.H. 1992. When it's hot, it's hot... or maybe not (Surface flaming may not portend extensive soil heating). *International Journal of Wildland Fire*, 2, 139-144.

HEINEMEYER, A. & VALLACK, H.W. 2012. *Literature review on: potential techniques to address heather dominance and help support 'active' Sphagnum supporting peatland vegetation on blanket peatlands and identify practical management options for experimental testing*. York: University of York draft report to Defra and Natural England.

HENDERSON, I.G., Fuller, R.J., Conway, G.J. & Gough, S.J. 2004. Evidence for declines in populations of grassland-associated birds in marginal upland areas of Britain. *Bird Study*, 51, 12–19.

HEWSON, R. 1982. The effect upon field vole (*Microtus agrestis*) habitat on removing sheep from moorland in west Scotland. *Journal of Zoology*, 197, 304-307.

HOBBS, R.J. & GIMINGHAM, C.H. 1987. Vegetation, fire and herbivore interactions in heathland. *Advances in Ecological Research*, 16, 870-193.

HOGG, E.H., LIEFFERS, V.J. & WEIN, R.W. 1992. Potential carbon losses from peat profiles - effects of temperature, drought cycles, and fire. *Ecological Applications*, 2, 298-306.

HOLDEN, J., Kirkby, M.J., Lane, S.N., Milledge, D.G., Brookes, C.J., Holden, V. & McDonald, A.T. 2008. Overland flow velocity and roughness properties in peatlands. *Water Resources Research*, 44, 11.

HOPE, D., PICOZZI, N., CATT, D.C. & MOSS, R. 1996. Effects of reducing sheep grazing in the Scottish Highlands. *Journal of Range Management*, 49, 301-310.

HUDSON, P. & NEWBORN, D. 1995. *A manual of red grouse and moorland management,* Fordingbridge: Game Conservancy.

IMESON, A.C. 1971. Heather burning and soil erosion on the North Yorkshire Moors. *Journal of Applied Ecology,* 8, 537-542.

INGHAM, H.A.P. 1983. Hydrology. In: *Mires, swamp, bog, fen and moor*. Vol. 4A. Ecosystems of the world. Amsterdam: Elsevier.

IUCN. 2011. *Burning and peatbogs*. Edinburgh: IUCN UK Peatland Programme.

JERRAM, R. & DREWITT, A. 1998. Assessing vegetation condition in the English uplands. *English Nature Research Report*, 264*.* Peterborough: English Nature.

JNCC. 2004. *Common Standards Monitoring guidance for upland habitats*. First Version 2004. Peterborough: Joint Nature Conservation Committee.

JOHNSON, E.A., Miyanishi, K. 2008. Testing the assumptions of chronosequences in succession. *Ecology Letters* 11, 419-431.

JONES, H..E., FORREST, G.I. & GORE, A.J.P. 1971. First stage for a model for the growth and decay of Calluna vulgaris at Moor House. In: Heal, 0.W. (ed.). *Tundra biome: working meeting on analysis and ecosystems, Kevo, Finland, September 1970*. London: IBP.

KEELEY, J.E. 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. *International Journal of Wildland Fire*, 18, 116-126.

KUHRY, P. 1994. The role of fire in the development of *Sphagnum*-dominated peatlands in western boreal Canada. *Journal of Ecology,* 82, 899-910.

LABATZ, J., Allott, T., Evans, M., Butcher, D., Billett, M., Stainer, S., Yallop, A., Jones, P., Innerdale, M., Harmon, n., Maher, K., Bradbury, R., Mount, D., O'Brien, H.& Hart, R. 2011. *Peatland hydrology*. Edinburgh: report to the IUCN UK Peatland Programme.

LANCE, A.N. 1983. Performance of sheep on unburned and serially burned blanket bog in western Ireland. *Journal of Applied Ecology*, 20, 767-776.

LINDSAY, R. 2010. *Peatbogs and carbon: a critical synthesis*. Sandy: University of East London report to the RSPB.

LINDSAY, R.A., Charman, D.J., Everingham, F., O'Reilly, R.M., Palmer, M.A., Rowell, T.A. & Stroud, D.A. 1988. *The Flow Country - The peatlands of Caithness and Sutherland. Edinburgh*: Peterborough: JNCC.

MACDONALD, A. 1999. Fire in the uplands: a historical perspective. *Scottish Natural Heritage.* Battleby.

MACDONALD, A.J., KIRKPATRICK, A.H., HESTER, A.J. & SYDES, C. 1995. Regeneration by natural layering of heather (*Calluna vulgaris*): frequency and characteristics in upland Britain. *Journal of Applied Ecology,* 32, 85-99.

MADDOCK, A. (ed.). 2008. *UK Biodiversity Action Plan: priority habitat descriptions*. BRIG.

MAFF. 1984. *The Heather and Grass Burning Code: advice on the burning of heather, grass, bracken , gorse and Vaccinium*. London: MAFF.

MAFF. 1986. *The Heather and Grass Burning Code: advice on the burning of heather, grass, bracken , gorse and Vaccinium*. 1986 version (with minor revisions). London: MAFF.

MAFF. 1992. *The Heather and Grass Burning code*. London: MAFF.

MAFF. 1994. *The Heather and Grass Burning code*. 1994 edition (with minor revisions). London: MAFF.

MALLON, D., WHEELER, P., WHITELEY, D. & YALDON, D.W. 2003. Mountain hares in the Peak District. *Brit. Wildlife*, 15, 110–116.

MALTBY, E., LEGG, C. J. & PROCTOR, M. C. F. 1990. The ecology of severe moorland fire on the North York Moors: effects of the 1976 fires, and subsequent surface and vegetation development. *Journal of Ecology,* 78, 490-518.

MARRS, R.H., RAWES, M., ROBINSON, J.S. & POPPITT, S.D. 1986. *Long-term studies of vegetation change at Moor House NNR: guide to recording methods and the database*. Merlewood: ITE.

MARTIN, D., FRASER, M.D., PAKEMAN, R.J. & MOFFAT, A.M. 2013. *Natural England Review of Upland Evidence 2013 - Review Five: impact of moorland grazing and stocking rates*. Sheffield: Natural England.

MCKENZIE, D., GEDALOF, Z.E., PETERSON, D.L. & MOTE, P. 2004. Climate change, wildfire and conservation. *Conservation Biology*, 18, 890-902.

MCMORROW, J., AYLEN, J., ALBERTSON, K., CAVAN, G., LINDLEY, S, HANDLEY, J & KAROONI, R. 2006. *Climate change and the visitor economy: challenges and opportunities for England's Northwest*. Oxford: Sustainability Northwest, Manchester and UKCIP.

MCMORROW, J.M. & LINDLEY, S.J. 2007. *Modelling the spatial risk of moorland wildfire*. Edale: University of Manchester report to Moors for the Future and Peak District Fire Operations Group.

MCVEAN, D.N. & LOCKIE, J.D. 1969. *Ecology and land use in upland Scotland*. Edinburgh: Edinburgh University Press.

MCVEAN, D.N. & RATCLIFFE, D. A. 1962. *Plant communities of the Scottish Highlands*. London: HMSO.

MILLER, G.R., JENKINS, D.& WATSON, A. 1966. Heather performance and grose populations. 1. Visual estimates of heather performance. *Journal of Applied Ecology*, 3, 313-326.

MILLER, G.R., MILES, J. & HEAL, O.W. 1984. *Moorland management: a study of Exmoor*. Cambridge: ITE.

MONTEITH, D.T. & EVANS, C.D. 2005. The United Kingdom Acid Waters Monitoring Network: a review of the first 15 years and introduction to the special issue. *Environmental Pollution*, 137, 3-13.

MOORE, P.D. 1973. The influence of prehistoric cultures upon the initiation and spread of blanket bog in upland Wales. *Nature*, 241, 350–353.

MOORE, P.D. 1975. Origin of blanket mires. *Nature*, 256, 267–269.

MOORLAND WORKING GROUP. 1998. *Good practice for grouse moor management*. Battleby: Scottish Natural Heritage.

NWCG. 2012. *Glossary of wildland fire terminology*. National Wildfire Coordinating Group.

NATURAL ENGLAND. 2008. *State of the natural environment 2008*. Sheffield: Natural England.

NCC. 1982. *Conservation of peatbogs*. London: Nature Conservancy Council.

NEARY, D.G., KLOPATEK, C.C. DEBANO, L.F. & FFOLLIOTT, P.F. 1999. Fire effects on belowground sustainability: a review and synthesis. *Forest Ecology and Management*, 122, 51-71.

NEWSON, S.E., CRICK, H.Q.P., CLARK, J.A. & MOSS, D. 2007. *Timing of breeding of moorland birds in Wales*. *BTO Research Report*, 362. Thetford: BTO.

NISBET, A. & GLAVES, D.J. 2009. Moorland management in Higher Level Stewardship: the evidence base for sustainable stocking rates. *Aspects of Applied Biology*, 100, *Agri-environment schemes – What have they achievedand where do we go from here?*, pp 141-149.

NT. 2005. *Heather burning: position statement as at February 2005*. Hope Valley: National Trust High Peak Estate.

NT. 2012. *High Peak moors vision and plan 2013-2038. Consultation draft*. Hope Valley: National Trust High Peak Estate.O'BRIEN, H., LABADZ, J.C. & BUTCHER, D.P. 2007. *Review of blanket bog management and restoration*. London: Nottingham Trent University research report to Defra.

OKLAND, R.H. 1990. A phytoecological study of the mire Northern Kisselbergmosen, SE Norway. 3: Diversity and habitat niche relationships. *Nordic Journal of Botany*, 10, 191-220.

OFFER, D., EDWARD, M. & EDGAR, P. 2003. *Grazing heathland: a guide to impact assessment for insects and reptiles*. English Nature Research Report, 497. Peterborough: English Nature.

PAGE, C.N. 1997. *The Ferns of Britain and Ireland*. Cambridge: Cambridge University Press.

PHILLIPS, J. 2012. *Moorland management: for agriculture, conservation and field sports: A pictoral guide*. Shrewsbury: Quiller.

PHILLIPS, J., WATSON, A. & MACDONALD, A. 1993. *A muirburn code*. Edinburgh: Scottish Natural Heritage.

PICKETT, S.T.A. 1989. Space for time substitution as an alternative for long-term studies. In: Likens, G.E. (ed.) *Long-term studies in ecology*. Chichester: Wiley.

PRESTON, C.D., PEARMAN, D.A. & DINES, T.D. 2002. The New Atlas of the British Flora. Oxford: Oxford University Press.

PYNE, S.J., ANDREWS, P.L. & LAVEN, R.D. 1996. *Introduction to wildland fire*. Chichester: Wiley.

RATCLIFFE, D.A. 1964. Mires and bogs. In: Burnett, J.H. (ed.) *The vegetation of Scotland*, pp. 426-478. Edinburgh: Oliver & Boyd.

RATCLIFFE, D.A. 1977. *A nature conservation review: the selection of biological sites of national importance*. Cambridge: Cambridge University Press for the nature Conservancy Council.

RATCLIFFE, D.A. 2002. *Lakeland*. London: Collins New Naturalist.

RATCLIFFE, D.A. & THOMPSON, D.B.A. 1998. The British uplands: their ecological character and international significance. In: Usher, M.B. & Thompson, D.B.A. (eds.) *Ecological change in the uplands*, pp 9-36. Oxford: Blackwell.

REDPATH, S.M., THIRGOOD, S., & CLARKE, R. 2002. Field vole Microtus agrestris abundance and hen harrier Circus cyaneus diet and breeding in Scotland. *Ibis*, 144, 1 on-line, E33-E38.

REIN, G., CLEAVER, N., ASHTON, C., PIRONI, P. & TORERO, J.L. 2008. The severity of smouldering peat fires and damage to the forest soil. *Catena,* 74, 304-309.

ROBERTS, J.L. & BOWMAN, N. 1986. Diet and ecology of short-eared owls *Asio flammeus* breeding on heather moor. *Bird Study*, 33, 12-17.

ROBICHAUD, P.R., BEYERS, J.L. & NEARY, D.G. 2000. *Evaluating the effectiveness of postfire rehabilitation treatments*. USDA Forest Service, Rocky Mountain Research Station, General Technical Report, RMRS-GTR-63.

ROBSON, G. 1998. The breeding ecology of curlew *Numenius arquata* on North Pennine moorland. Sunderland: PhD thesis, University of Sunderland.

RODWELL, J. 1991. *et seq*. *British plant communities. Vols.1-5.* Cambridge, Cambridge: Cambridge University Press.

RODWELL, J. 1991. *British plant communities. Vol. 2: Mires and heaths,* Cambridge: Cambridge University Press.

ROSENBURGH, A. & MARRS, R. 2010. *The heather beetle: a review*. Liverpool University report to the Heather Trust.

ROWE, J.S. 1983. Concepts of fire effects on plant individuals and species. In '(Eds RW Wein, R.W., MacLean, D.A. (eds.) *The role of fire in northern circumpolar ecosystems*, pp 135–154. Chichester: Wiley.

SCOTTISH GOVERNMENT. 2011a. *The muirburn code: a guide to good practice*. Revised version. Edinburgh: The Scottish Government.

SCOTTISH GOVERNMENT. 2011b. *Prescribed burning on moorland. Supplement to the Muirburn Code: A guide to best practice*. Edinburgh: The Scottish Government.

SEERAD. 2001a. *The muirburn code: a guide to good practice*. Edinburgh: SEERAD.

SEERAD. 2001b. *Prescribed burning on moorland: supplement to the Muirburn Code: a guide to good practice*. Edinburgh: SEERAD.

SHEPHERD, M. J., LABADZ, J., CAPORN, S. J., CROWLE, A., GOODISON, R., REBANE, M. & WATERS, R. 2013. *Natural England review of upland evidence - Restoration of Degraded Blanket Bog*. Natural England Evidence Review, Number 003.

SHERRY, J. 2005. *Guidelines for burning and cutting dwarf-shrub vegetation on lowland and upland heath and blanket bog*. Bangor: Countryside Council for Wales.

SIM, I.M.W., GREGORY, R.D., HANCOCK, M.H. & BROWN, A.F. 2005. Recent changes in the abundance of British upland breeding birds. *Bird Study*, 52, 261-275.

SIMMONS, I.G. 2003. *The moorlands of England and Wales*. Edinburgh: Edinburgh University Press.

SIMMS, C. 1969. Recolonisation of burnt heath by lizards *Lacerta vivipara* Jacquin. *British Journal of Herpetology*, 4, 665-676.

SIMMS, C. 1972. Shift in a population of northern vipers. *British Journal of Herpetology*, 4, 268-271.

SLACK, N.G. 1990. Bryophytes and ecological niche theory. *Botanical Journal of the Linnean Society*, 104, 187-213.

SLATER, F.M. & AGNEW, A.D.Q. 1977. Observations on a peat bogs ability to withstand public pressure. *Biological Conservation*, 11, 21-27.

SMITH, P., SMITH, J., FLYNN, H., KILLHAM, K., RANGEL-CASTRO, I., FOEREID, B., AITKENHEAD, M., CHAPMAN, S., TOWERS, W., BELL, J., LUMSDON, D., MILNE, R., THOMSON, A., SIMMONS, I., SKIBA, U., REYNOLDS, B., EVANS, C., FROGBROOK, Z., BRADLEY, A. & FALLOON, P. 2007. *ECOSSE: Estimating Carbon in Organic Soils – Sequestration and Emissions*. Edinburgh: Scottish Executive Environment and Rural Affairs Department.

SOLOMON, S., QIN, D., MANNING, M., MARQUIS, M., AVERYT, K., TIGOR, M.M.B., LEROY MILLER, H. & CHEN, Z. 2007. *Climate change 2007: The physical science basis*. Contribution of working group 1 to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Geneva: IPCC.

STODDART, D.M. & HEWSON, R. 1984. Mountain hare, *Lepus timidus*, bags and moor management. Journal *of Zoology*, 204, 563-565.

STONE, D.A. 2013. *Natural England Evidence Reviews: guidance on the development process and methods (1st Edition 2013)*. Natural England Evidence Review, Number 001.

STROUD, D.A., CHAMBERS, D., COOK, S., BUXTON, N., FRASER, B., CLEMENT, P., LEWIS, P., MCLEAN, I.,BAKER, H. & WHITEHEAD, S. 2001. *The UK SPA network: its scope and content*. 3 Vols. Peterborough: JNCC.

SWT. 1995. *Conservation management of blanket bog : a review.* Sandy: Scottish Wildlife Trust report to the RSPB.

SYDES, C. & MILLER, G.R.1988. Range management and nature conservation in the British uplands. In: Usher, M.B. & Thompson, D.B.A. (eds.) *Ecological change in the uplands*. Special publication 7 of the British Ecological Society. Oxford: Blackwell Scientific Publications.

TALLIS, J.H. 1969. The blanket bog vegetation of the Berwyn Mountains, North Wales. *Journal of Ecology*, 57, 765-787.

THOMPSON, D.B.A., MACDONALD, A.J. & HUSDSON, P.J. 1995a. Upland moors and heaths. *In:* SUTHERLAND, W. J. & HILL, D.A. (eds.) *Managing habitats for conservation*, pp 292-326*.* Cambridge: Cambridge University Press.

THOMPSON, D.B A., GILLINGS, S.D., GALBRAITH, C.A., REDPATH, S.M. & DREWITT, J. 1997. The contribution of game management to biodiversity: a review of the importance of grouse moors for upland birds. In: Fleming, L.V., Newton, A.C., Vickery, J.A. & Usher, M.B. (eds.) *Biodiversity in Scotland: status, trends and initiatives*. Edinburgh: The Stationary Office.

THURMAN, E.M. 1985. *Organic geochemistry of natural waters*. New York: Kluwer Academic Publishers Group.

UK BIODIVERSITY GROUP. 1999. *Tranche 2 Action Plans: Vol. VI – terrestrial and freshwater species and habitats*. Peterborough: English Nature on behalf of the UK Biodiversity Group.

UNITED UTILITIES. 2012. *United utilities' response to Natural England's review of upland evidence*. United Utilities. [Submitted to the UER by United Utilities].

USHER, M. B. & THOMPSON, D. B. A. 1993. Variation in the upland heathlands of Great Britain: conservation importance. *Biological Conservation,* 66, 69-81.

VILLAGE, A. 1987. *The kestrel*. Calton: Poyser.

VIITANAN, P. 1967. Hibernation and seasonal movements of the viper, *Vipera berus berus* (L.), in southern Finland. *Annales Zoologici Fennici*, 4, 472–546.

WAG 2008. *The Heather and Grass Burning code for Wales 2008*. Cardiff: Welsh Assembly Government.

WALKER, L.R., WARDLE, D.A., BARDGETT, R.D., CLARKSON, B.D. 2010. The use of chronosequences in studies of ecological succession and soil development. *Journal of Ecology*, 98, 725-736.

WARD, S.D. 1972. The management of vegetation by swaling. *Devon Trust for Nature Conservation Journal,* 4, 186-190.

WARD, S.D., JONES, A.D. & MANTON, N. 1972. The vegetation of Dartmoor. *Field Studies*, 3, 505-533.

WATT, A.S. 1955. Bracken versus heather, a study in plant sociology. *Journal of Ecology*, 43, 490-506.

WEBB, J.R., DREWITT, A.L. & MEASURES, G.H. 2010. Managing for species: Integrating the needs of England's priority species into habitat management. Part 1 Report. *Natural England Research Report*, NERR024*.* Sheffield: Natural England.

WEIN, R.W. 1973. Biological flora of the British Isles: *Eriophorum vaginatum*. *Journal of Ecology*, 61, 601-615.

WHEELER, P. 2008. Effects of sheep grazing on abundance and predators of field vole (*Microtus agrestis*) in upland Britain. *Agriculture, Ecosystems & Environment*, 123, 49-55.

WHITELEY, D. 1997. *Reptiles and amphibians of the Sheffied area and north Derbyshire*. Sorby Record Special Series, 11. Sheffield: Sorby Natural History Society and Sheffield City Museum.

WHITELEY, D. 2003. Early records of adders in the Peak District. *Sorby Record*, 39, 67.

WHITTAKER, E. 1960. Ecological effects of moor burning. Aberdeen: PhD thesis, University of Aberdeen.

WHITTINGHAM, M.J., PERCIVAL, S.M. & BROWN, A.F. 2000. Time budgets and foraging of breeding golden plover Pluvialis apricaria. *Journal of Applied Ecology*, 37, 632-646.

WHITTINGHAM, M.J., PERCIVAL, S.M. & BROWN, A.F. 2001. Habitat selection by golden plover *Pluvialis apricaria* chicks. *Basic and Applied Ecology*, 2, 177-191.

WHITTINGHAM, M.J., PERCIVAL, S.M. & BROWN, A.F. 2002. Nest-site selection by golden plover: why do shorebirds avoid nesting on slopes? *Journal of Avian Biology*, 33, 184-190.

WIGGINTON, M.J. 1999. *British Red Data Books. 1. Vascular plants*. 3rd Edition. Peterborough: JNCC.

WILD, C. & ENTWISTLE, C. 1997. Habitat management and conservation of the adder in Britain. *British Wildlife*, 8, 287–295.

WORRALL, F., CHAPMAN, P., HOLDEN, J., EVANS, C., ARTZ, R., SMITH, P. & GRAYSON, R. 2010b. Peatlands and climate change. Report to IUCN UK Peatland Programme. Edinburgh.

WORRALL, F., ROWSON, J.G., EVANS, M.G., PAWSON, R., DANIELS, S. & BONN, A. 2011b. Carbon fluxes from eroding peatlands - the carbon benefit of revegetation following wildfire. *Earth Surface Processes and Landforms,* 36, 1487-1498.

WORRALL, F. & EVANS, T.P., ROWSON, J.G., WARBURTON, J. & ADAMSON, J.K. 2009. The multi-annual carbon budget of a peat-covered catchment. *Science of the Total Environment*, 407, 4084-4094.

WOTTON, S., LANGSTON, R. & GREGORY, R. 2002. The breeding status of the Ring Ouzel *Turdus torquatus* in the UK in 1999. *Bird Study*, 49, 26-34.

YALLOP, A.R., CLUTTERBUCK, B. & THACKER, J.I. 2009. Burning issues: the history and ecology of managed fires in the uplands. *In:* BONN, A., ALLOTT, B., HUBACEK, K. & STEWART, J. (eds.) *Drivers of Environmental change in uplands.* London: Routledge.

YALLOP, A.R., CLUTTERBUCK, B. & THACKER, J.I. 2011. Changes in water colour between 1986 and 2006 in the headwaters of the River Nidd, Yorkshire, UK: a critique of methodological approaches and measurement of burning management. *Biogeochemistry*, 111, 97-103.

YEO, M.J.M. 1997. Blanket mire degradation in Wales. In: Tallis, J.H., Meade, R. & Hulme, P.D. (eds) *Blanket mire degradation: causes, consequences and challenges*, pp. 101-115. Aberdeen: Maccaulay Land Use Research Institute for the British Ecological Society.

Appendix 1 Summary of evaluated studies

Table A Summary of evaluated studies

Study/main reference	Date	Main sub-question	Other sub-question(s)	Type/ quality	Country	Region/area[1]
ADAS 1997	1997	h		2++	England	PD
Albertson *et al*.	2010	g		2+	England	PD
Allen	1964	d		2++		
Allen *et al*.	2013	c		2+	England	PD
Amar *et al*.	2011	b		2++	UK	
Anderson *et al*.	2006	a	f	4+	Wales	
Anderson *et al*.	2009	h		2-	UK	
Armstrong *et al*.	2009	d		3-	England	NP
Armstrong *et al*.	2012	d		3+	England	NP
Aspray	2012	d		2++	England	NP,YD,PD
Aylen *et al*.	2007	g		4-	England	PD
Beharry-Borg	2009	d		2+	England	YD
Benscoter *et al*.	2011	e		1+		
Brown & Bainbridge	1990	b		4-	UK	
Burch	2008	a		2-	England	NYM
Chambers *et al*.	2007	a		3+	Wales	
Chapman *et al*.	2010	d		2-	England	YD
Chapman *et al*.	2009	a		2-	England	PD
Chen *et al*.	2008	c		2++	England	NP (MH)
Clay & Worrall	2011	c		2+	England	PD
Clay *et al*.	2010b	c		2++	England	NP (MH)
Clay *et al*.	2012	d		2+	England	Northum
Clutterbuck & Yallop	2010	d		2++	England	SP,YD
Cotton & Hale	1994	a		1-	England	SP
Coulson	1988	b		2-	England	North
Coulson *et al*.	1992	a		4+	UK	
Couwenberg *et al*.	2011	c		2-	Belarus	

Table continued…

Study/main reference	Date	Main sub-question	Other sub-question(s)	Type/ quality	Country	Region/area[1]
Critchley *et al*.	2011a	h		2++	England	
Critchley *et al*.	2011b	h		2++	England	
Currall	1981	a	f	2+	Scotland	
Curtis & Corrigan	1990	b		2-	Scotland	
Daplyn & Ewald	2006	b		2-	England	PD
Davies	2005	e		2+	Scotland	
Davies & Legg	2008	a		2+	Scotland	
Davies & Legg	2011	e		2+	Scotland	
Davies *et al*.	2008	g		4+	UK	
Davies *et al*.	2010	e		2+	Scotland	
Dennis & Eales	1997	b		2+	England	Northum
Elliott	1953	a		2-	England	PD
Ellis	2008	a		3+	Scotland	
Eyre *et al*.	2003	b		2+	Scotland	
Farage *et al*.	2009	c		2-	England	YD
Fletcher *et al*.	2010	b		1+	England	Northum
Forrest & Smith	1975	a		2-	England	NP (MH)
Fullen	1983	c		3+	England	NYM
Garnett *et al*.	2000	c		1+	England	NP (MH)
Garnett *et al*.	2001	c		2++	England	NP (MH)
Glaves *et al*.	2005	b		4+	UK	
Grand-Clement	2008	c		2-	England	NP (MH),PD
Grant	2012	b		4+	UK	
Grant *et al*.	2012	b		4+	UK	
Gray & Levy	2009	c		4+	UK	
Grayson *et al*.	2012	d		2+	England	SP,PD
Hamilton	2000	a		2-	Scotland	
Hard Hill vegetation	1965-2001	a	f	1++	England	NP (MH)
Hard Hill hydrology	2005-08	d	f	1+	England	NP (MH)
Harris *et al*.	2006	a		2+	England	PD
Harris *et al*.	2011a	e	c	2+	England	PD
Harris *et al*.	2011b	a		2+	England	PD
Haworth & Thompson	1990	b		2+	England	SP

Table continued…

Study/main reference	Date	Main sub-question	Other sub-question(s)	Type/ quality	Country	Region/area[1]
Hochkirch & Adorf	2007	b		2++	Germany	
Holden	2005a	c		2+	UK	
Holden *et al*.	2012	d		2++	UK	
Holden *et al*.	2013	d		1+	England	PD,NP
Holmes *et al*.	1993	b		2-	Wales	
IUCN	2011	a		4+	UK	
JNCC	2009	a		4-	UK	
Jones	2005	a		4+	Wales	
Kinako & Gimmingham	1980	c		2+	Scotland	
Legg & Davies	2009	e		4+	Scotland	
Lindsay	2010	c		2+	UK	
Lindsay & Ross	1994	a		2++	England	Cumbria
Lindsay *et al*.	1988	e		4+	UK	
Littlewood *et al*.	2011	a		4-	UK	
Loftus	1994	a		2-	Ireland	
Lunt *et al*.	2011	a		4-	UK	
MacDonald	2008	a		2+	Scotland	
Marrs *et al*.	2004	a	f	1+	England	PD,YD
McDonald *et al*.	1991	d		2++	England	
McFerran *et al*.	1995	a	b	2-	Scotland	
McMorrow *et al*.	2009	g		2+	UK	PD+
Miles	1971	a	f	1+	Scotland	
Miller	2008	d		2++		
Mitchell & McDonald	1995	d		2-	England	YD
Moss *et al*.	2005	b		2++	England	
Mowforth & Sydes	1989	a		4-	UK	
Natural Engalnd	2009	h		2+	England	
O'Brien *et al*.	2005	d		2-	England	PD
O'Reilly	2008	a		2-/4+	England	Northum
Orwin & Ostle	2012	c		1+	England	NP (MH)
Penny Anderson Associates	2012	h		2++	England	
Pattison & Lane	2011	d		4+	USA	
Pearce-Higgins & Grant	2006	b		2+	UK	

Table continued…

Study/main reference	Date	Main sub-question	Other sub-question(s)	Type/ quality	Country	Region/area[1]
Pearce-Higgins *et al.*	2009	b		4+	UK	
Picozzi	1968	b		2+	Scotland	
Pietikainen *et al*.	1999	c		2+	Finland	
Ramchunder *et al*.	2009	d		2+	England	North
Ramchunder *et al*.	2013	d		2++	England	Pennines
Ratcliffe	1990	b		4+	UK	
Rawes & Williams	1973	a	f	2+	England	NP (MH)
Ross *et al*.	2003	a	f	1+	England	Northum
Rowell	1980	a		4-	UK	
Shaw *et al*.	1996	a	f	4+	UK	
Shelter *et al*.	2008	c		2+	US	
Smith *et al*.	2001	b		2+	UK	
Sotherton *et al*.	2009	b		4+	UK	
Stewart *et al*.	2004	a		1++	UK/Ireland	
Stone	2006	a	b	2-	England	PD
Taylor & Marks	1971	a	f	1+	England	NP (MH)
Tharme *et al*.	2001	b		2+	Scotland /England	
Thompson *et al*.	1995b	a		4+	UK	
Tucker	2003	a	b,f	2+/4+	UK	
Usher	1992	b		2-	England	NYM
Ward *et al*.	2007	a	c,d,f	1++	England	NP (MH)
Worrall & Warburton	2009	d		2+	England	NP
Worrall *et al*.	2011a	a-d		4+	UK	
Worrall *et al*.	2010a	c		2+	UK	
Worrall *et al*.	2012	d		2-	England	PD
Yallop & Clutterbuck	2009	d		2++	England	SW,PD,NYM
Yallop *et al*.	2005	h		2+	England	
Yallop *et al*.	2008	d		2+	England	YD
Yallop *et al*.	2010	d		2++	England	SP
Yallop *et al*.	2012	h		2++	England	Inc. NP,PD,NYM
Yallop *et al*.	2006a	h		2+	England	NP

[1] NP = North Pennines (MH = Moor House), Northum = Northumberland, NYM = North York Moors, PD = Peak District, SP = South Pennines, SW = South West and YD =Yorkshire Dales

Appendix 2 Summary of current Codes of Practice, guidance and reviews providing context for the topic review

The Burning Regulations and Code

Burning is regulated in England by the Heather and Grass etc Burning (England) Regulations 2007. The Regulations impose restrictions on the burning of heather, rough grass, bracken, gorse and *Vaccinium* which include the dates and timing of burning, the precautions that must be in place. No burning is allowed between 15 April and 1 October in the uplands (SDA). These Regulations introduced new prohibitions of burning in certain situations that "may create a high risk of soil exposure and erosion." Of particular relevance to blanket peat, this included: burning so as to expose bare soil extending more than 25 m along the bank of a watercourse (including grips) and not leaving soil smouldering for more than 48 hours.

The Heather and Grass Burning Code for England (Defra 2007) includes "peat bog and wet heathland... (including blanket bogs, raised bogs, valley bogs or mires, springs and flushes)" in the list of sensitive areas where "there should be a strong presumption against burning", although this is qualified by "other than in line with a management plan agreed with Natural England" and that "such plans are likely to involve careful burning on long rotations, with cool burns leaving large amounts of "stick" and not damaging the moss layer." Later in the code guidance is given on rotations, including: "on deep peat (ie more than 50 cm/20 in deep) aim to burn on long rotations of 15-25 years. The associated Best practice guide on identifying sensitive areas (Defra 2008a) clarifies that peat bog and wet heathland includes modified dwarf shrub-dominated forms. Previous versions of the Code simply state: "do not burn peat bog and wet moor (flow ground)" (MAFF 1992, 1994); and that "burning should in principle be avoided, and certainly restricted to the absolute minimum necessary to meet the needs of stock" (MAFF 1984, 1986).

Similar codes have been produced in Wales and Scotland. The Welsh Code (WAG 2008) has a similar presumption that "sensitive habitats and areas" including "peat bog and wet heath including raised, blanket and valley bogs and mires in upland and lowland situations … should be included in no-burn areas unless burning of them is part of an agreed restoration or other environmental management programme." The Scottish Muirburn Code (SEERAD 2001a; Scottish Government 2011a) adopts a slightly different approach. "Blanket bogs and raised bogs on deep peat (more than 0.5 m - about 20 inches - deep)" are included in a list of situations where burning should not be carried out ('fire free areas') unless *Calluna* constitutes more than 75% of the vegetation cover.

Other good practice guidance

There is a wide range of other sources of guidance and recommendations on good and best burning practice, particularly regarding moorland including upland peatland habitats. This tends to recommend that burning on mire habitats, particularly blanket bog, and sometimes wet heath, should be minimised, for example, avoiding particularly vulnerable areas or burning only on long rotations (Phillips *et al.* 1981, 1993, Anderson 1986; Mowforth & Sydes 1989; Usher & Thompson 1993; Shaw *et al.* 1996). Other guidance suggests that these habitats should generally not be burnt except in specific circumstances (NCC 1982, Rowell 1988, 1990, MAFF 1992a, Coulson *et al.* 1992, Brown 1995, SWT 1995, Jerram & Drewitt 1998, Hudson & Newborn 2005, Thompson *et al.* 1995a, Backshall *et al.* 2001, Sherry 2005 and Glaves *et al.* 2005). However, much of this guidance does not provide definitions of blanket bog and other mire habitats, leaving some uncertainty over which areas it relates to on the ground. Similar guidance generally recommending not burning on blanket bog/deep peat has also been produced for some local geographical areas, for example, for the

south-west moors in general (Defra in press a) and specifically for Dartmoor (NCC 1989, Bates 1993), but also including the National Trust's High Peak Estate (NT 2005, 2013) and United Utilities' land holdings in the Pennines and Bowland (United Utilities 2012).

Agri-environment schemes and designated site consents

Agri-environment schemes (AES) in England generally deal with burning on moorland with a requirement to agree a burning plan tailored to the site. This has varied in the approach taken to burning on blanket bog and wet heath. There is also a requirement to follow the burning code unless practice is modified in the agreement. The Moorland management template for SSSI grouse moors agreed between the then English Nature and the Moorland Association states that "there is particular concern to see only very careful burning on areas of blanket bog (that is, vegetation that generally covers deeper peat) to avoid damaging the peat resource as well as for maintaining or restoring the delicate surface vegetation". In many areas burning of blanket bog is either not needed or is not desirable for nature conservation and/or wider land management purposes. Where heather is currently dominant on blanket bog, burning management may still be appropriate as part of a conservation management/restoration plan. However, where burning takes place it will always be important to ensure that the peat resource and any sensitive vegetation are not damaged" with similar text for wet heath. Away from grouse moors guidance tends to recommend not burning on blanket bog, for example, for Dartmoor ESA. Thus, there are inconsistencies nationally in how burning on blanket bog is treated in AES and on SSSIs.

Burning effects and previous reviews

The effects of burning on blanket bog and other upland peatlands depend on various aspects of the fire regime including the frequency and severity of burning along with other factors including: time since previous burning; post-burn environmental conditions; exposure and altitude; starting composition and condition; water levels; and intensity of livestock grazing and trampling (for example, Whittaker 1960; Gimingham 1972; Tucker 2003). Thus, whilst the effects of any particular burn can be very variable, the scale, pattern and severity, particularly of repeated, rotational, burning at a larger-scale are also important in determining the effects on biodiversity and ecosystem services.

There have been a series of previous reviews that are relevant to the topic of burning on peatlands, mostly in terms of the effect on biodiversity although some have been wider: Mowforth & Sydes (1989) and Coulson *et al.* (1992) covering moorland management in general; Shaw *et al.* (1996) covering burning and grazing management specifically of blanket bog and wet heath; Tucker (2003) covering impacts of burning in the uplands on soils, hydrology and biodiversity; and Stewart *et al.* (2004) a rigorous, systematic 'evidence-based' procedure to analyse published information on the impact of burning on blanket bog (and separately upland heath). In addition, Defra established an independent science panel to advise on their then review of the Heather and Grass Burning Regulations and Code (Glaves *et al.* 2005). These reviews have generally concentrated mainly on the impacts on biodiversity, especially on vegetation composition and structure, but more recently have increasingly covered wider impacts including the important ecosystem services peatlands provide. However, the wider impacts on carbon balance and water quality were already recognised earlier, for example, by Coulson *et al.* (1992). Nevertheless, there has been a major increase in research and publications on ecosystem services particularly the ability of blanket bogs to sequester carbon and on increasing water colouration/DOC in peatland drainage waters including some specifically relating to the role of controlled burning. This continues unabated reflecting the growing interest in ecosystem services and, in particular, the water utilities' direct interest in water quality and especially colouration. The impact of burning on peatlands, particularly blanket bog, has recently been reviewed as part of the International Union for Conservation of Nature (IUCN) UK Peatland Programme's Commission of Inquiry[8]. The Inquiry commissioned a series of technical reviews including one specifically on the Impacts of burning management on peatlands (Worrall *et al.* 2011).

[8] See: **www.iucn-uk-peatlandprogramme.org/commission**

They also published a separate briefing paper on burning (IUCN 2011 part of the text of which also appears in the overall inquiry report: Bain *et al.* 2011) which presents the IUCN's review of the evidence drawing on the technical review and other, in some cases more recent, evidence including a review on water colouration by Holden *et al.* (2011). Some of the other IUCN technical reviews also refer to burning impacts: peatland restoration (Lunt *et al.* 2011), peatland hydrology (Labadz *et al.* 2011), peatland biodiversity (Littlewood *et al.* 2011). These reviews were evaluated in the current topic review.

Appendix 3 Studies and evidence for the effects of burning on vegetation composition, structure and function

Hard Hill plots vegetation studies

Hard Hill burning and grazing experiment vegetation study, 1954-present [1++][9] at Moor House NNR in the North Pennies. The experiment, including the block/plot layout and treatments, which has more recently been additionally utilised for hydrology and soils monitoring (see 'Hard Hill plots hydrology and water chemistry studies' – Appendix 7), is described in detail by Marrs *et al.* (1986) and Holden *et al.* (2012). The plots lie in an extensive area of high altitude, less-modified, active *Calluna-Eriophorum* blanket bog (NVC community M19 with a tendency to M20 post-burning) with high cover of *Eriophorum vaginatum* and *Calluna*, and relatively frequent *Sphagnum* spp. Most has not been burnt now for at least 60 years and escaped the worst of the aerial pollution associated with industrial pollution that more severely affected the South Pennines.

The experiment is thought to be the only long-term experimental investigation of burn rotations on blanket bog in the UK and, as such, is described in some detail below. Large grazing exclosures were established in 1954 in a partially randomised block design of four blocks at differing altitudes up a slope x 2 grazing treatments (grazed, ungrazed) x 3 burning treatments (with burning only randomised within a grazing treatment for practicality) in 30 m x 30 m plots. The wider area was grazed by sheep at a low overall rate of 0.1/ha with no winter grazing. Burning of the main plots was at 10 and 20 year intervals following burning of all the plots in 1954, with the control plots not burnt after 1954 (59 years ago). The years that burning was carried out differed slightly from those planned due to unsuitable conditions in some years: 1954/55 (all), 1965 (10-year), 1975 (10- and 20-year), 1984 (10-year), 1995 (10- and 20-year) and 2007 (10-year) (after the last plot vegetation monitoring in 2001). Thus, since the initial burn, the 10-year treatments have had four further burns and the 20-year two over the period of vegetation monitoring to date (up to 2001).

Vegetation surveys were carried out (year since last burn in brackets for short (S) and long (L) rotations) in: 1961 (6-7 year S and L), 1972/73 (6-7 year S, 16-17 year L), 1982 (7 year S and L), 1991 (7 year S and 16 year L) and 2001 (6 year S and L) and a further resurvey is overdue. Thus, apart a partial resurvey in 1965 (0 year S, 9-10 year L) and early post-burn studies between 1976-80 (see below), most of the surveys have been carried out 6-7 year after the previous burn (for all five surveys of the short rotation plots and three of the long rotation plots surveys) apart from two surveys, in 1972/73 and 1991, when the 20-year plots were 16-17 year.

In addition to the main experiment plots, 'reference plots' were also established outside of the main blocks which were thought to have been unburned for at least 30 year prior to 1954, though these were only surveyed in 1965 and 2001.

The effects of burning and grazing in the vegetation of the Hard Hill plots are described in an extensive series of papers and reports: Rawes & Williams (1973), Rawes & Hobbs (1979), Hobbs & Gimmingham (1980), Hobbs (1981), Hobbs (1984), Adamson & Kahl (2003)/Adamson pers. comm. (2004) to Stewart *et al.* (2004) and Lee *et al.* (accepted), which are described separately below although inevitably there is some overlap between them. Data from two of the papers, Rawes & Hobbs (1979) and Adamson & Kahl (2003)/Adamson pers. comm. were also included in the

[9] Note that this large, long-term experiment, and the numerous publications on it, was treated and evaluated as a single study in terms of the vegetation component. This is consistent with other studies with several publications (see paragraph 3.16).

systematic review by Stewart *et al.* 2004. The findings from the publications from this study are described below split by hydrology and water quality, and water chemistry (but as they are regarded as part of a single study are not given separate study type/quality classes).

Rawes & Williams (1973) reviewed the production and utilisation of *Calluna* and *Eriophorum* at Moor House NNR in relation to red grouse and livestock grazing. As part of this, they summarised some of the results from the Hard Hill plots between 1961 and 1972. They noted that the cover of *Calluna* is very variable from site to site (at Moor House) "due not only to previous a management and present stocking; but also to aspect, moisture and degree of slope." It was suggested that "long rotation burning without sheep grazing produces a good quantity of *Calluna* shoots; but as a management, burning and sheep grazing together are likely to be unsuitable" partly due to an increase in bare ground. They concluded that "if sheep farming is the main land use, increased sheep grazing will produce better summer feed; but if grouse shooting is of foremost importance, either light grazing and no heather burning, or burning without sheep grazing are recommended." In addition to their description of the Hard Hill plots experiment, Rawes & Williams (1973) and Rawes & Hobbs (1979) describe a separate sheep-grazing trial carried out at Moor House that included a burning treatment which is described separately below.

Rawes & Hobbs (1979) summarised the effects in the Hard Hill experiment between 1961 (6-7 year after the first burn) and 1972/73 (6-7 year since the last burn in the 10-year treatments and 18-19 year in the 20-year treatments). After seven year (1961), there were no significant differences in plant cover and height of the main vascular plant species between plots (not yet re-burned) or grazing treatments, though bare peat was more frequent in the grazed plots. After 18-19 year (1972), there were significant differences for some species following the (10-year) burn in 1965, especially in the grazed treatment: *Calluna* declined, but *Rubus chamaemorus*, *Eriophorum* spp., rusty swan-neck moss *Campylopus flexuosus*, tumid notchwort *Lophozia ventricosa* and nodding thread-moss *Pohlia nutans* increased. There were also significant changes in structure with the burned, grazed plots having the least dense vegetation and shortest vegetation and bare ground increased. An additional survey of key species in 1976 one year after a burn showed better vegetative regrowth and seedling establishment of *Calluna* in the 10-year than 20-year burn treatments. It was suggested that "at that stage in the experiment it could be said that burning stimulates the productivity of many plants. This may be due to the fertilising effect of ash, or to better light penetration as a result of removal of the canopy. However, the situation is by no means stabilised after 22 years."

Hobbs & Gimmingham (1980) summarised the data from the Hard Hill experiment mainly on the basis of continuation of additional resurveys of key species in 1977-78 following those described by Rawes & Hobbs 1979 (1-3 year after the last burn for both burning treatments). This indicated that the differences noted between the short and long rotations by Rawes & Hobbs (1979, above) were reduced the following year and by the third year there were no significant differences in *Calluna* vegetative regrowth, whilst the frequency (in 10 cm x 10 cm quadrats) was considerably higher in the long rotation plots as was the frequency of most bryophytes and lichens. *Rubus chamaemorus*, on the other hand was more frequent in the short rotation plots. "It thus appears that frequent burning produces a temporary increase in the availability of young *Calluna* shoots, but that this is short-lived. After the initial increase, the short rotation output is reduced to levels similar to, or possibly lower than, that of the long rotation. Ecologically, the vegetation is in a steady-state without burning (Forrest 1971), and if productivity is little improved, a case could be made that burning us an unnecessary management option for the wetter heaths where the problem of heather degeneracy does not occur. Certainly, frequent burning shifts the system or less permanently to *Eriophorum* dominance, in much the same way as grazing does. This process is analogous to that in the *Molinia*-dominated communities...."

Hobbs (1984, 1981) reported on the effects in the Hard Hill plots up to 1980 comprising: the 1961 and 1972 surveys (also reported by Rawes & Hobbs 1979); additional data from transects surveyed annually between 1978 and 1980; and *Calluna* performance between 1976 and 1980. In 1961 (6-7 year after the initial burn of all plots), not surprisingly no distinct differences in species abundances were detected between treatments, but there were significant differences between blocks and distinct differences between replicates, with low cover of *Calluna* in one block, whilst cover of *Eriophorum*

angustifolium, *Rubus chamaemorus* and *Sphagnum* species varied greatly between blocks. These spatial differences were still apparent in 1972/73 (6-7 year after burning in the 10-year plots and 18-19 year in the 20-year plots) when some differences between burning treatments became apparent. *Calluna* was considerably more abundant in the (not yet re-burnt) long-rotation plots as might be expected and some bryophytes including *Campylopus flexuosus* and *Lophozia ventricosa* were more abundant in the short-rotation plots. By 1978-80 (24-26 year after the first burn and 3-5 year since the last burn for both burn treatments) *Eriophorum vaginatum* was dominant in the burnt plots, with *E. angustifolium* co-dominant with, in many cases, the two forming a dense, closed mat. The results indicated that burning favoured *Calluna* only initially, ie in the very short-term (1-2 year), because of the subsequent and rapid re-growth of *Eriophorum vaginatum*, which came to dominate for c.10 year. The author suggested that burning with the objective of improving *Calluna* availability is unlikely to be appropriate to high altitude *Calluna-Eriophorum vaginatum* blanket mire because *Calluna* appears to achieve a 'steady state' naturally through the continuous layering and rejuvenation of stems among the *Sphagnum* carpet. Instead, repeated burning is likely to favour dominance by *Eriophorum* which is preferred by sheep during summer.

Adamson & Kahl (2003) provided a brief overview of the long-term results of burning and grazing in the Hard Hill plots up to 2001. Light grazing was shown to have very little effect on vegetation composition, which was attributed to the absence of stock during winter when dwarf shrubs are targeted preferentially. "*Eriophorum* dominated the vegetation in the initial post-burn period" with "*Calluna* taking 11-17 years to reach its maximum abundance" (though other references suggest a continued increase beyond this). Although slower initially, recovery of *Calluna* was then more rapid with a burn repeat time of 20 year compared to 10 year. *Calluna* rejuvenated successfully in the absence of burning because shoots layered within the *Sphagnum* carpet. *Rubus chamaemorus* was favoured by no grazing and burning, attributed to highly selective grazing by sheep and improved light intensities in post burn plots, respectively. It was noted that "without fire, *Calluna* was found to naturally reach a "steady state" in which it constantly "rejuvenated" by shoot layering in the *Sphagnum* carpet, indicating that fire makes a questionable management tool at this altitude even for grouse."

Lee *et al.* (2013) reported on the results from the surveys using pin frames in 1972/73, 1982, 1992 and 2001 (over a 28 year period up to 47-48 year since the initial burn) and also a resurvey of the separate but adjacent 'reference plots' in 2011 following the only previous survey in 1965 (ie after 46 year after c.90+ year unburnt). In the main experiment plots, species-richness/plot, bare peat and three species, showed both significant year and burn treatment effects. Species-richness was highest, and bare peat lowest, in the control plots (both $p < 0.001$), with the latter highest in the short rotation plots ($p < 0.01$), but both declined over time (both $p < 0.001$). *Calluna* showed an increasing trend overall over time ($p < 0.05$) in all treatments, with highest cover in the control plots and lowest in the short rotation plots. *Eriophorum vaginatum* and *Campylopus flexuosus* both declined overall over time (both $p < 0.001$), with highest cover in the short rotation plots and lowest in the controls. Four variables showed weakly significant differences between burn treatments and no year effect: number of moss spp., number of *Sphagnum* species and frequency of both S. *capillifolium* (probably ssp. rubellum) and all *Sphagnum* species. The *Sphagnum* -related variables were highest in the 10-year burn plots (all $p < 0.05$) and were overwhelmingly related to one species, S. *capillifolium*, with 677 pin hits across all plots/surveys c.f. just 12 for all other species. The number of moss species/plot was lower in the long rotation c.f. the control plots ($p < 0.05$). A further three species showed significant year x burn-rotation interactions: *Eriophorum angustifolium* increased with increasing burning frequency and both heath plait-moss *Hypnum jutlandicum* and hart's-tongue thyme-moss Plagiomnium undulatum increased in the control plots (all $p < 0.001$). At the community level, there was an increasing difference over time between the burn treatments and control with grazing having an insignificant effect.

It was concluded that the evidence "suggests that leaving blanket bog on a very long prescribed burning rotation at this site will increase the *Calluna* vulgaris-*Hypnum jutlandicum* component in a similar manner to the reference plots. Where a short-rotation prescribed burning program is used the vegetation will shift towards one dominated by peat-forming *Eriophorum* spp. and *Sphagnum* here" although *Sphagnum* did not attain dominance with only one species at all frequent. The dominance of

E. vaginatum following burning is consistent with earlier findings from the experiment and of graminoid-dominance elsewhere.

The resurvey of the unburned (since c.1924) reference plots showed some significant changes in the 46 year between 1965 and 2011 even after at least 87 years since burning. Eight species were only recorded in 1965 (five lichens and three liverworts), but 24 were new in 2011, (16 mosses, six liverworts and single lichens and vascular plants) including some species at relatively high frequency: *Hypnum jutlandicum*, red-stemmed feater-moss *Pleurozium schreberi*, little shaggy-moss *Rhytidiaelphus loreus*, *Sphagnum fallax* and Muller's pouchwort *Calypogia muelleriana*; and the locally scarce Lesser Twayblade *Listera cordata*. Only one *Sphagnum* species was recorded in 1965, *S. capillifolium rubellum* (the only frequent species recorded in the main experiment plots), but a further six were recorded in 2011, although only *S. fallax* was frequent. This might suggest that a greater diversity of *Sphagnum* species and change in dominance might take a considerable period after burning on this site and are yet to occur in the main experiment plots. Overall species richness (SR) did not change significantly, though moss SR increased significantly, with seven new species, whilst lichen SR declined, with five species lost including three *Cladonia* species. Only three species recorded in both years showed a significant change in cover between years, with *Calluna*, bog bead-moss *Aulacomnium palustre* and *Plagiothecium undulatum* all increasing in cover. Notably, *Calluna* doubled in cover (to 55.3%), but not to overwhelming dominance and, unlike in some other studies on more severely modified, drier blanket bog (for example, Harris *et al.* 2011b), this did not seem to be at the expense of the other species present.

Other Hard Hill plots vegetation studies

Ward *et al.* (2007) [1+] sampled vegetation composition in their separate, wider study of carbon dynamics (described further under sub-question c) at the long-term Hard Hill plots. This study only used the 10-year burn and 'unburned' since 1954 treatments for both grazed and ungrazed areas from all four blocks in 2003-04, at the end of the 10-year burn cycle. Vegetation community composition was determined by measuring the quantity (g dry weight) of live above ground shoots for each species present from an area of 25 cm^2 within each plot at quarterly intervals between April 2004 and May 2005. Results were subsequently grouped in to three broad types: graminoids, ericoid sub-shrubs and bryophytes (not separating out Sphagna).

Changes in the relative contribution of the three plant groups were observed due to burning and grazing, with the greatest effects being due to burning. Burning increased the biomass of graminoids by 88% relative to unburned controls ($p = 0.009$), but reduced the biomass of shrubs and bryophytes by 51% ($p < 0.0001$) and 92% ($p < 0.0001$), respectively. The changes due to grazing were similar, but of smaller magnitude. These findings are consistent with the observations from vegetation composition monitoring, but perhaps larger in magnitude (reflecting biomass rather than cover).

Taylor & Marks (1971)/Marks & Taylor (1972) [1+] used one of the four replicate blocks at Hard Hill to study the effect of burning and grazing on the growth of *Rubus chamaemorus*. It is found strictly in *Calluna-Eriophorum* blanket mire but with a restricted geographic distribution attributed to climate, but the factors affecting the production of flowers and fruit not fully understood. The authors counted shoot and fruit densities and harvested the aerial biomass of *Rubus chamaemorus* on four occasions between June and August 1969 in ten 1 m^2 quadrats located at random in plots subject to 10 and 20 year burn rotations, either with or without sheep grazing at a low stocking rate. The last burn for the 10 year rotation was 1965, four years before the study period so the results may have missed initial changes, for example, prompted by the immediate nutrient flush. Aerial biomass was harvested and segregated into leaves and stems plus petioles (plus flowers), for which total dry weight, N, P, K, Ca and Mg contents were determined. The total numbers of flowers and fruits present per plot was enumerated on each sampling occasion. The rhizome and root fraction was separated from a single core sample of 0.08 m diameter by 0.5 m depth in August 1969 and the same suite of parameters was quantified. The ratio of shoot:root biomass was determined per plot and used to estimate each parameter per m^2. In a related experiment to discriminate the effects of canopy removal and nutrient addition following burning twenty x 1 m^2 samples within the 20 year burn rotation plots were clipped of all biomass down to the bog surface in April 1969. Bulked biomass was incinerated and the

resulting ash was applied to 10 of the clipped plots, selected at random. The density of shoots and fruits produced in each 1 m^2 was monitored 3 and 15 months following treatment.

The studies showed that the density of shoots, total above ground biomass and rhizome biomass was highest in the 10 year burn rotation plots and much more so in the absence of grazing. However, individual shoot weights were similar between 10 year and 20 year burn rotations subject to grazing, where shoot density tended to drop between mid-July and August. Grazing caused a proliferation of smaller shoots with a lower propensity to flower and set fruit. There were no differences among treatments in the concentrations in aerial dry matter of total N, P, K, Ca and Mg, so that total nutrient content per m^2 mirrored aerial biomass. The addition of inorganic nutrients to clipped vegetation was far less influential than grazing in constraining the density of shoots and flowers: plants in fenced plots accrued an even higher density of both in the second season after burning. The authors concluded that the ability of *Rubus chamaemorus* to maintain internal nutrient concentrations in plots where plants are released from grazing and able to proliferate vigorously demonstrates that nutrients do not limit its growth. Furthermore, sheep appear to seek out *Rubus chamaemorus*. They suggest that managed burning will not therefore benefit this or other plant species whose distribution and abundance is controlled more by climate and/or synecological relationships.

Other vegetation studies

Elliott (1953) [2-] studied the effects of burning on seven *Calluna*-dominated moorland stands in the Peak District. They generally occurred on shallow peat and varied in composition, trending to degraded wet heath, although the vegetation was characteristically species-poor (suggested as the consequence of long term historic burn management) with only ten species occurred consistently in the stands. Most sites had been subject to systematic managed burning whilst some had fallen victim to ‘accidental’ fire. The age of each burn was estimated by determining growth rings in *Calluna* stems.

The main changes in response to burning were: an initial decline in *Empetrum nigrum* sometimes leading loss under regular burning; an initial increase in cowberry *Vaccinium myrtillus*, for which regular burning was detrimental and a shorter cycle lead to progressive loss; a reduction in cowberry *Vaccinium vitis-idaea* and its eventual elimination under a regime of repeated burning; and a temporary increase in *Deschampsia flexuosa* up to approximately 6 year after burning, after which it started to decline. Data from one of the wetter stands (Ringinglow Bog) were analysed in the systematic review by Stewart *et al.* (2004, below). They described the site as M15 wet heath and summarised the changes as: a minimal increase in bare ground; a decline in *Empetrum nigrum*, *Vaccinium myrtillus* and *V. vitis-idaea*; an increase in *Calluna*; and a large decline in *Deschampsia flexuosa*, similar to those across the other sites. Elliott also noted evidence for the loss of *Listera cordata*, a species indicative of *Sphagnum* -rich bog vegetation with an open *Calluna* canopy.

Miles (1971) [1+] investigated the effect of burning a mosaic of *Trichophorum-Eriophorum* blanket bog with abundant *Molinia* and *Molinia*-dominated grassland on grazing utilisation by red deer on the island of Rhum, Scotland. Sample plots in bog and grassland with abundant *Molinia* were burned and grazing rates and the accumulation of *Molinia* dry matter compared with adjacent unburned ground.

Grazing was significantly greater in burned than unburned *Molinia*-dominated areas in the spring and summer following burning. Grazing rates declined again in both the two years after burning. Burning reduced litter by 80% on grassland and 85% on bog, but within two years the quantity of litter had increased to half that on unburned areas. No changes in vascular plant cover were evident either due to burning or to excluding grazing. Burning killed most of the *Sphagnum* moss but this had mostly recovered after two years. New growth of *Molinia* on burned ground was attractive to grazing deer and burning therefore increased grazing value but such value was evident only in the year of burning and therefore increased the need to burn to maintain grazing value. The benefits of burning were short-lived. Alternatives to burning were considered; on *Molinia* grassland heavy grazing may encourage more palatable grasses such as *Agrostis* and *Festuca*. On *Molinia*-rich bog it was suggested that intensive grazing could prompt change to species-poor *Juncus squarrosus* bog. The

author suggested that burning *Molinia*-rich vegetation is unlikely to result in benefits as it perpetuates *Molinia* dominance.

Forrest & Smith (1975)/Forrest (1971) [2-] studied the variation in annual production of a range of less-modified, active blanket bog types at Moor House NNR over a three-year period. The study sites comprised five unburnt sites and two (assumed conventional management) burnt sites (burnt c.9 years previously) on deep peat.

Total annual net production varied between sites ($p < 0.001$) from 481-868 g m^{-2} y^{-1} and between-year variation was relatively small compared to that between sites. Mean total (above and below ground) production was highest on the burnt sites (805 +/- 64 g m^{-2} y^{-1}) than on the unburnt *Calluneto-Eriophoretum* (629 +/- 54 g m^{-2} y^{-1}) and *Trichophoro- Eriophoretum* (491 g m^{-2} y^{-1}) sites. It was not clear to what extent the high production on these burnt sites was due to burning versus differences in baseline community types. The study was not designed to examine changes in floristic composition resulting from burning, but species frequencies on burnt and unburnt sites were reported (from 50 25-cm square quadrats per site). The two burnt sites had higher frequencies of a number of species, including *Eriophorum angustifolium*, *Trichophorum cespitosum* and bog asphodel *Narcthecium ossifragum*, together with a number of *Sphagnum* species, while the frequency of some species, such as *Empetrum nigrum* and the lichens *Parmelia saxatilis* and *Cladonia squamosa* were lower. Data from the study were analysed as part of the systematic review by Stewart *et al.* (2004, below) who interpreted the changes 9 year after the burn as: *Eriophorum angustifolium* becoming dominant; *Empetrum nigrum*, *Rubus chamaemorus* and *Vaccinium oxycoccus* declining; *Trichophorum cespitosus* increasing; and many bryophytes and lichens declining, but *Sphagnum papillosum* and *S. recurvum* increasing.

Rawes & Williams (1973)/Rawes & Hobbs (1979) [2+] described an unreplicated sheep-grazing trial carried out at Moor House NNR (separate from the Hard Hill experiment, above) that included a burning followed by heavy (3.4 sheep/ha) and light (0.37 sheep/ha) grazing treatments, and an ungrazed control plot (but no burnt, ungrazed plot). Botanical composition was surveyed annually using the pin-frame method used at the Hard Hill plots.

Rawes & Hobbs (1979) summarised the results after eight years. Under heavy grazing the initial botanical changes were similar in the burnt and unburnt plots: a rapid loss of *Calluna* and quick recovery of *Eriophorum vaginatum*. The cover of mosses was reduced in the heavy grazing plots, but liverworts showed a contradictory response as did *Rubus chamaemorus* probably reflecting differences in initial, patchy distributions. Lichens declined under heavy grazing and bare ground increased and after 7 year was at 11.4% and 8.6% in the burnt and unburnt heavy grazed plots. Heavy grazing with and without burning completely changed the structure of the vegetation to a shorter sward reflecting the loss of *Calluna*. Thus, in the initial stages there was little evidence that burning produced a difference compared to the unburnt, heavily grazed treatment.

Currall (1981)/Currall (1989) [2+] studied the effect of management burning on wet heath vegetation on the island of Skye in western Scotland in 1977-78, principally by a post-burn chronosequence across 53 stands of varying ages after burns up to a >20 year category. In addition, the effects of grazing, clipping and raking were investigated using grazing exclosures. The objectives were to study the use and control of fire; vegetation responses to fire; interactions of grazing and burning; and alternative methods to achieve the objectives of burning.

The author suggested that post fire successions in wet heath in NW Scotland typically follows three phases. Firstly, there is a 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 ericoids, mat-grass *Nardus* and total bryophytes decline significantly, though species-richness increases initially then gradually declines. The second phase is a ‘dense graminoid phase’ and results from the establishment of dense growth of *Molinia* or *Trichophorum cespitosum* and a reduction in bare ground, though *Erica* tetralix may peak in this phase. This seems to be characteristic of wet heath, not normally being seen in dry heath successions. The phase may last 8-12 year. *Calluna* and other ericoids tend to become dominant in the third phase, typically c.15

year after the fire, while graminoids decline and bryophytes develop under the canopy. With further time, species such as Potentilla erecta and *Eriophorum vaginatum* may reappear as gaps occur in the heather canopy. It was suggested that burning on short rotations and/or heavy grazing after burning can lead to maintenance of the dense graminoid phase and hence dominance of *Molinia* and *Trichophorum* and reduction in *Calluna*.

Cotton & Hale (1994)/Hale & Cotton (1988) [1-] carried out an experimental, randomised control trial on Ilkley Moor in the South Pennines to assess the effectiveness of two cutting treatments as alternatives to traditional moorland burning practice. They set up fifteen rectangular plots (each approximately 70 m^2) on moorland dominated by *Calluna*, *Empetrum nigrum* and *Vaccinium myrtillus* on shallow peat (described by Stewart *et al.* 2004 as M15 wet heath). Each plot was assigned to one of three management treatments: burning, flailing (with litter left on the soil surface) and rolling back the vegetation (by using spades to cut major stems and above-ground material, which was then manually rolled to the sides of the plot) with untreated controls, all left open to grazing. Plots were monitored for ten years.

Flailed plots developed new vegetation cover with as much *Calluna* as burned plots, but more slowly (lagging behind burned plots by 1 year). Plots where the vegetation was cut and rolled back did not revegetate as successfully. The frequency of *Empetrum* did not change significantly over time in any of the treatments. Most of the regeneration on the burning and rolling sites was from seed, whereas flailing encouraged vegetative regrowth. The study concluded that, where management by burning is not practicable, flailing would be a viable alternative technique to manage *Calluna* for grazing and grouse. The papers do not directly compare the results for treatment plots with the non-treatment plots, but Stewart *et al.* (2004, below) analysed the Hale & Cotton (1988) data in their systematic review (see below). They summarised the changes from five year post-experimental-burning as showing a large increase in bare ground, a decline in the dwarf shrubs *Empetrum* (contra comment in Cotton & Hale 1994 over longer ten-year period) and *Vaccinium myrtillus*, resulting in a decline in ericoid frequency and diversity, but only minor other changes in floristic composition.

Lindsay & Ross (1994)/Lindsay (1977) [2++] studied the effects of an extensive, severe wildfire in 1976 on vegetation at Glasson Moss, a previously less modified lowland raised bog NNR which is one of the sites in the South Solway Mosses NNR in Cumbria. Though a lowland site, it was included in the review as it provides important information on *Sphagnum* recovery after fire on a high quality, *Sphagnum* -rich site. Lindsay (1977) mapped the extend of the burn and vegetation types and sampled each with quadrats recording, in particular, *Sphagnum* presence, cover and damage from the fire, in 1977 flowing a previous survey before the fire in 1975. Lindsay & Ross (1994) describe the results of regular monitoring of recovery up to c.15 year following the burn using fixed transects across the unburnt and burnt areas along which vegetation and surface topography were recorded and stereo photos taken.

In 1977, other than the narrow unburnt strip of vegetation, no evidently living *Sphagnum* remained on the site although in many areas it was possible to find hummocks or lawns which had been severely singed by the fire rather than utterly destroyed. It was not possible to say at that time whether such damaged *Sphagnum* was alive or dead. By 1979, a few small pockets of evidently living *Sphagnum* had appeared and much of the singed material still remained but then subsequently decomposed. This was followed by an initial flush of *S. tenellum*, a coloniser species, reaching peak cover within five year but then declining, followed by slow re-appearance of typical bog Sphagna. This was accompanied by a similar recovery of the fire-damaged surface, changing from a relatively flat condition lacking any evident pattern to a markedly hummock-hollow topography.

Loftus (1994) [2-], as described in Stewart *et al.* (2004, see below), studied the impact of burning in the presence of sheep grazing on *Calluna-Eriophorum* (M19) blanket bog (condition and degree of modification unknown) in the Wicklow Mountains, Ireland. The cover of all species was recorded using pin frames in a site comparison of three recently burnt (3 year post-burn) with three unburnt but grazed sites.

Though it proved not possible to obtain the original thesis for this review, Stewart *et al.* (2004, below) analysed the data in their systematic review and summarised the results as showing “minor changes in floristic composition including minor changes in key CSM table species.” This was interpreted as representing no change in condition.

McFerran at al. (1995) [2-] studied the composition and structure of *Calluna*-dominated moorland (and effects on invertebrates) in response to burning and grazing, at Ballycastle, in County Antrim, Northern Ireland, to compare post-fire succession with published studies for *Calluna* elsewhere. The site was common grazing (entirely hill sheep present for the 75% of the year) but shooting rights, exercised historically for red grouse, were in single-ownership. Three areas of moorland were defined, deemed to represent ‘space for time’: unburnt vegetation; vegetation burnt in 1982 when an extensive wildfire affected 30 ha; and areas of c30 m x 60 m subject to managed burning in autumn 1988. Three replicate monitoring plots of 30 m x 15 m were established in each of the burn types; the effect of grazing was also investigated by erecting three replicate grazing exclosures in each. The percentage cover of species and the actual and relative canopy contribution in seven contiguous 5 cm height strata was determined by point quadrats in autumn 1989 and 1990 (up to 2 and 8 year after management burning). Vegetation canopy height and proportional contribution of species to above ground biomass was assessed for unburnt and 1982 burns only, in autumn 1989 and 1990. The percentage frequency of species in the newly burnt stands was monitored monthly in permanent quadrats for a period of 18 months.

The areas burnt in a wildfire 1982 had a higher proportion of live to dead *Calluna* biomass than the unburnt stands ($p < 0.001$), where more litter had accumulated ($0.5 > p < 0.001$) and the canopy height is greater ($p < 0.001$). An increase in above ground biomass, canopy height and contribution of live *Calluna* expected on release from grazing was detected after a period of 12 months ($p < 0.05$ for each). These trends were also reflected in differences in percentage cover among the three burn types. The percentage cover of *Vaccinium myrtillus* was highest in the most recently burnt areas ($p < 0.05$). Sedge *Carex* species were most prevalent in the 1982 managed burn plots. Live *Calluna* biomass was distributed higher within the vegetation canopy in unburnt compared to both burnt areas ($p < 0.01$). Burning was shown to initiate complex successional pathways which have characteristics species associations. Data from the study were analysed in the systematic review by Stewart *et al.* (2004) who summarised the initial changes 1 year after burning as: a minimal increase in bare ground; and a decline in *Calluna*, *Eriophorum vaginatum* and bryophytes.

Hamilton (2000) [2-] investigated fuel load and fire characteristics in blanket bog vegetation in the NW Highlands of Scotland. The recovery of *Sphagnum* was also investigated in relation to fuel load and ‘fire intensity’. The vegetation studied was predominantly M17 blanket bog with some M18 bog and M15 wet heath. The study sites were mainly dictated by the location of management burns carried out by land managers but were chosen so as to be typical and for proximity to roads for ease of access. Within sample locations sample plots were located randomly along 40 m or 2 x 20 m transects. Sample plots were paired to allow grazed and ungrazed treatments (the latter protected by cages). In 1996 four sites were used to record vascular plant regrowth and recovery of the *Sphagnum* layer after fire. These sites were first visited between 2 and 6 days after they were burnt. In 1997 three sites were set up primarily to record maximum fire temperatures, fuel load, and effects of increased fuel load, but loss of samples and data meant that the only data available recorded the recovery of the *Sphagnum* layer. In 1998 experimental burns were used to record the temperature regime of fires and quantify fire intensity. Five sites were used, with some plots being harvested (to develop fuel loading equations) and others being burnt. *Sphagnum* recovery was also recorded on these sites.

It was suggested that the study demonstrated that simple and accurate equations to predict different fuel types can be developed for fuels in the blanket-bog habitat. The results from these equations to predict pre-fire loading emphasise the variability in fuels across the habitat, and thus the need for more than `spot' sampling of fuels, as has been the general practice in UK fire studies. The fire characteristics results (temperature, intensity) were broadly comparable to those obtained from previous studies in heathland habitats in the UK and results confirm that variability is a key feature of all fires, but in particular those in blanket bog. The main reason for fire variability is the spatial

patchiness of the fuel complex, and to a lesser degree the extreme moisture gradients encountered in the fuels, for example, in relation to the topography of the bog surface. Experimental burns included addition of fuel too half of the individual flat plots dried birch twigs, up to a maximum of 0.5 cm in diameter. The results from all sites showed a large variation in the effects of fire on *Sphagnum* condition between sites. The amount of *Calluna* (as measured by a score value and height) has a significant effect on a condition score of the *Sphagnum.* This may be due to differences in the character of fuel where there is tall *Calluna*, topography (*Calluna* on bogs is often associated with a hummock, resulting in drier fuel) or other factors. It is unlikely that fuel load alone is responsible for the differences in condition scores because no significant changes in *Sphagnum* condition score resulted from adding extra fuel. Significant relationships between the condition scores and fire temperature characteristics suggested that these are important factors in determining the damage caused to the *Sphagnum* layer by fire. Grazing appears to reduce *Sphagnum* recovery perhaps through trampling. Observations suggested that fire may aid *Sphagnum* regrowth in some situations – perhaps through removing litter which otherwise swamps/shades the moss surface.

Ross *et al.* (2003) [1+] used a split-plot design to evaluate management techniques for controlling *Molinia caerulea* and enhancing *Calluna vulgaris* in M15 wet heath in Northumberland. Burning was the core treatment of the study; it may be desirable from the point of view of rejuvenating *Calluna* but risky in terms of the propensity to disproportionately invigorate *Molinia* and other less-desirable species. The main plots of the design were heavy and light sheep grazing at 1.5 and 0.66 ewes/ha, respectively, which were prescribed for upland dwarf shrub heath under the former ESA scheme. Three replicate blocks of 28 m x 28 m were selected in each area, within which four treatments were allocated at random to 10 m x 10 m plots each provided with a 2 m buffer around the perimeter: control with no burning; burning only; burning followed by cutting to 8 cm; and burning followed by grass-specific herbicide application. Plots were burnt in April 1996, while cutting and spraying were undertaken in the following July. The vegetation was monitored initially in July 1995 prior to any treatments and annually thereafter in July from 1997-1999 (up to 3 year after the burn). Three variables were quantified in 100 10 cm x 10 cm sub-quadrats for five 1 m x 1 m permanent quadrats in each plot: % cover in the top layer of vegetation, determined by the cross-wire method; % frequency; and dominance, or the number of cells in which each species is the dominant component. Dominance data were not collected in 1997. Results from successive sampling occasions were used to derive the rate of change in dominance from *Calluna* to *Molinia*, *Deschampsia flexuosa* and common sedge *Carex nigra*, and from *Molinia* to *Calluna*. Results for the two main plots were analysed separately.

Three types of vegetation were identified: *Molinia*-dominated, *Calluna* (or *Calluna*-Eriophorum)-dominated and mixed heath. Subsequent changes in response to treatments were manifest as shifts of these groups within ordination space and related to the positions on the ordination of key plant species. ANOVA with multiple comparison tests was used to investigate differences in frequency and dominance among treatments and with time. Predictably, the frequency and dominance of *Calluna* was reduced in all three treatments and under both grazing rates, reflecting initial loss of biomass and subsequent slow rate of recovery following a burn. However, the frequency of *Molinia* was not altered significantly; burning with herbicide treatment showed a tendency to reduce *Molinia* in the following year but not thereafter, and the effect was only at the higher stocking rate. In the vegetation types where *Molinia* was prevalent, follow-up herbicide did reduce its dominance; for other treatments dominance increased. At the 10 x m c.10 cm scale, *Calluna* was replaced as the dominant species by *Molinia*, *Deschampsia* and *Carex*, depending on grazing rate: the grasses were favoured by light grazing and *Carex nigra* by heavier grazing. The results emphasise that initial vegetation composition influences changes in dominance among key species in wet heaths following burning, and that management of the grazing regime may be the most effective means of suppressing undesirable grasses and thereby favouring dwarf shrubs. Data from the study were analysed in the systematic review by Stewart *et al.* (2004) who summarised the changes 3 year after burning as: a decline in *Calluna* and an increase in *Molinia* grazed at both stocking rates.

Marrs *et al.* (2004) [1+] conducted a six-year factorial RCT experiment to test a range of management treatments designed to reduce *Molinia* and encourage the development of dwarf-shrub vegetation at sites in the Peak District and Yorkshire Dales. In each region two sites, one on species-

poor ‘white’ *Molinia*-dominated moor and another on ‘grey’ moor where *Molinia* occurs with dwarf shrubs, were selected (soil types not given). A nested factorial design was applied with a burned v unburned treatment; year round sheep grazing; summer grazing and ungrazed control; and two rates of herbicide (glyphosate) treatment with an untreated control. A supplementary experiment with litter removal and *Calluna* seed application was used to assess methods of *Calluna* re-establishment on ‘white’ moors.

Burning had relatively little effect on community composition and reductions in sward height following burning were followed by rapid recovery within two years. Grazing tended to shift the community of ‘grey’ moors towards bog-moorland types (not well defined in the study) but had little effect on ‘white’ moors. Grazing levels in the study were low and differences between summer only and year round treatments were not seen in practice due to stocking patterns. At the Dales ‘grey’ site herbicide treatment tended to shift vegetation towards acid grassland, but elsewhere herbicide had little effect on vegetation composition. Treatment to increase *Calluna* seedling density produced evidence of initial colonisation but seedling densities declined as *Molinia* recovered. There were marked variation in control of *Molinia* and subsequent response even in apparently similar vegetation types. In some cases a significant amount of time was needed to detect change. It was suggested that at ‘white’ sites where *Molinia* is dominant a package of management measures including initial *Molinia* control, surface treatment to disturb the litter layer and addition of seed will be required to restore dwarf-shrub cover. It was suggested that a range of communities should be considered as end-points; especially where *Molinia* forms mixtures with dwarf shrubs as treatment may cause a shift towards acid grassland and that once a mixed *Molinia*–*Calluna* vegetation is established appropriate management by grazing and burning should be applied.

Stone (2006) [2-] investigated vegetation and invertebrate communities on blanket bog in the Peak District (described further under sub-question b).

Cover of *Calluna* increased to a maximum in the mature phase then declined. There was a negative correlation between grass abundance and heather cover. *Vaccinium myrtillus* occurred in young stands but was generally absent from mature and degenerate stands and cross-leaved heath occurred only where there were gaps in the heather canopy in pioneer and degenerate phases.

Harris *et al.* (2006) [2+] in a pilot study examined the effect of managed cool burning on *Calluna*-dominated, severely modified degraded blanket bog community composition in two geographically distinct moors in the Peak District where ‘cool burning’ had been used for the previous 15 year. A chronosequence of burns was defined on aerial maps in 4 x 1 km^2 replicate areas at Bamford and Howden Moors, from which ten were selected at random for each site. Within these, ten 0 1 m x 1 m quadrats were located randomly and the percentage cover of all plant species was recorded along with bare ground, litter, dung and a range of simple environmental variables (slope, aspect, vegetation height, soil pH, soil C-content, altitude, *Calluna* growth form).

Classification of combined quadrat data (using TABLEFIT) to define NVC type at each site revealed M19 blanket mire at Bamford and M20 blanket mire at Howden. Quadrat data were explored initially using DECORANA, which showed some separation of the sites and that all environmental variables had a significant relationship with axis scores. Further analysis using constrained RDA analysis showed elevation and elapse time post-burning to be significant, and orthogonal, explanatory variables but much weaker than other unknown factors. HOF models were successful in identifying a subset of species that increased with elapse time (*Calluna vulgaris* and *Deschampsia flexuosa*) and others that decreased (for example, dwarf swan-neck moss *Campylopus pyriformis* and *Eriophorum vaginatum*), as well as some showing relationships with elevation (for example, *Rubus chamaemorus*). It was suggested that cool burning was achieving, if slowly, its goal of rejuvenating *Calluna* but that community composition was strongly influenced by a complex combination of environmental factors including grazing.

Burch (2008, 2009) [2-] examined the regeneration of bryophytes in burns of differing elapse time within wet (and dry) heath (NVC communities M16d (and H12a)) from a chronosequence on Spaunton Moor in the North York Moors, to explore the suitability of vascular plant canopy height as

a surrogate for optimal biomass accumulation in *Calluna* and a reliable criterion by which to initiate sustainable burning. The density of bryophyte shoots of different species was determined for eight replicate 10 cm^2 quadrats in burns of non-degenerate *Calluna* of the range 1, 3, 5, 7, 10, 15, 20 and 25+ years elapsed time, in three wet heath sites (NVC type M16d) (and four dry heath sites, NVC type H12a). The composition of the bryophyte community was assessed within the same burns by recording the presence of species every metre along 3 m x 10 m transects. Chronosequence transect data were classified (using TWINSPAN) to define communities of bryophytes and their possible relationship to elapse time since burning. Although the first dichotomy was meaningful for wet (and dry) heath, subsequent divisions could not be interpreted simply. The relationship between canopy height at different times after burning and shoot density was examined for selected species by Spearman's rank correlation; differences among *Calluna* growth phases were determined using Friedman's test.

The numbers of stems of heath star-moss *Campylopus introflexus* and *Sphagnum* species as a group were negatively correlated with vascular plant canopy height and greatest in canopies less than 30 cm. In contrast, the relationship was positive for *Hypnum jutlandicum* which was most abundant at canopy heights around 30 cm and declined once canopies reached 55 cm. Broadly similar patterns were observed for the wet heath sites. However, some species, including some *Sphagnum* species more typical of mire/wetland vegetation only occurred, or increased, in older stands. Stands of *Calluna* of different age and growth phase varied widely in vascular plant canopy height. Based on the extent to which the bryophyte flora was considered to have recovered, heights of and 25-30 cm on wet heath (and 41-54 cm for dry heath) were suggested as most suitable for burning to maintain bryophyte diversity and more reflective of optimal biomass accumulation than age or growth phase.

Davies & Legg (2008)/Davies (2001) [2+] examined the impact of regular management burning on the lichen diversity of *Calluna*-dominated vegetation. Lichen diversity was sampled at 26 fire sites which had been burnt three months to 18 year previously, as well as 11 long-term unburnt sites (not burnt for at least 25 years) in two separate study areas at Mar Lodge in the Scottish Highlands. Only study area 1 was on deep peat. Lichen richness was assessed in each burn site from only two 50 cm quadrats (used as separate data independent points).

General Linear Modelling demonstrated that there was a significant difference in the number and type of lichen species recorded in the two areas ($p < 0.001$). Generally, across both sites, terricolous lichens were less frequent where pleurocarpous mosses were more abundant ($p < 0.005$) and area 1 (wetter, on deep peat) had more pleurocarpous mosses and less terricolous lichen species. In area 1, terricolous lichen richness was heavily dependent on *Calluna* stand development; in burnt areas, species richness increased with *Calluna* height, but decreased in unburnt areas ($p < 0.001$). Corticolous (bark-living) species richness increased with *Calluna* stand height ($p < 0.001$). As the canopy opens up in the degenerate phase, mosses maintained their dominance, preventing terricolous species re-establishing, but the thick woody stems and more open canopy allowed for colonisation by corticolous lichens. Overall (using data from both sites), terricolous lichen diversity was low immediately after burning. but increased rapidly in subsequent years, reaching a peak between 10-20 years, before starting to decline. Corticolous species showed a steady increase in richness following burning, with a peak in the unburnt areas. The authors note that lichen susceptibility to fire is likely to vary depending on the species' growth habitat. Corticolous species are probably more affected by burning because they are more likely to be destroyed as the fire burns the *Calluna* canopy. Bryophillous lichens may be protected from fires by damp moss mats or micro-topography, while terricolous species are likely to be able to colonise bare ground rapidly. Species richness was highly variable in unburnt areas and is affected by other factors, including the extent of trampling and grazing. For example, in area 1, zones of higher lichen species were frequently found in building and mature heather areas that were crossed by deer tracks. It was concluded that whilst fire allows the formation of areas of young vegetation with high terricolous diversity, areas of older *Calluna* provide better habitat for corticolous species and that burn return intervals of less than 15-20 year are likely to lead to a decline in lichen diversity.

MacDonald (2008) [2+] as part of a review the literature relating to the potential impact of burning and/or compaction by machinery or animals on Scottish lowland raised bogs (see below), undertook a study of five raised bogs with a good land management history and an aerial photographic record stretching back several decades. From the photos, areas were identified with evidence of burning (and/or track construction), which were then assigned a relative date (for example, 'serially burned', 'long unburned'), and eight 1 x 1 m semi-randomly-located quadrats were placed, either paired with a quadrat outside the potential impact area, or unpaired, and a number of variables relating to vegetation composition, structure and previous treatments were recorded.

There were highly significant effects due to treatments and sites, but total *Sphagnum* cover was a less important discriminator than variables such as the total amount of dwarf-shrub cover, total graminoid cover, and total cover of 'shrubby' lichens. For paired quadrats, *Sphagnum* cover difference (serially burned minus long unburned or on-track minus off-track) between treatments was not significant ($p = 0.55$). But the effect of site on differences in *Sphagnum* cover was significant ($p = 0.007$). Unpaired quadrats showed a significant effect of treatment ($p = 0.02$), but not site ($p = 0.70$), on *Sphagnum* cover. There were significant differences between the long unburned areas and the serially burned areas only at one site ($p = 0.009$). There were also significant negative relationships between: *Sphagnum* cover and total cover of graminoids ($p = 0.001$), non-*Sphagnum* bryophytes ($p < 0.001$), 'shrubby' lichens ($p < 0.001$), surfaces covered by algae or crustose lichens ($p < 0.001$); and a weaker positive relationship with 'wetness' ($p = 0.009$). Overall the results indicated that highly significant differences between sites and treatments were little related to differences in *Sphagnum* cover. There was no significant relationship between *Sphagnum* cover and dwarf shrub (mainly *Calluna*) cover, although there were significant negative relationships between *Sphagnum* cover and cover of graminoids, other bryophytes, shrubby lichens, and crustose lichens. It was concluded that the results did not provide sufficient justification for routine burning (or compaction) as a restoration tool on raised bogs.

O'Reilly (2008) [2-] investigated the autecological characteristics of Sphagna in peatland communities in the North Pennines with the aim of identifying possible indicator species for rapid field assessment of the impact of management practices. Twenty 1 km^2 sampling plots were selected at random from known peatlands within the NA boundary within which the percentage cover of all plant species was recorded for 10 x 0.25 m^2 quadrats located using stratified random sampling based on a transect across mire. A wide range of environmental variables was also recorded in the field including: physical characteristics (vegetation height, slope, aspect, altitude); chemical characteristics (pH, conductivity); quantitative measures relating to the potential impact of management activities (for example, distance to nearest grip); and additional floristic data from the wider area to provide NVC type and the cover of peat-building Sphagna plus *Eriophorum vaginatum*. The main dataset was supplemented by an additional 41 quadrats from a separate study in which the concentration of a subset of extractable nutrients and heavy metals in the peat had been determined by laboratory analysis. The two datasets were analysed separately using DCA followed by CCA, with both full and most parsimonius models being generated for the latter. Trends for species abundance elucidated by multivariate analysis were investigated for individual environmental parameters using linear regression. Four derived measures were generated for the main dataset: overall plant species richness; *Sphagnum* species richness; and a 'Peat Building Index' (based on the cumulative cover of peat-building Sphagna plus *Eriophorum vaginatum*). As well as the two measures of richness, the Shannon Diversity index was also calculated for the 41-sample data-subset. Relationships between each of these community measures and potentially important environmental variables (as shown from DCA and CCA) including burning were investigated using generalised linear modelling (GLM) for continuous variables and non-parametric correlation (Kendall's Tau) for ordinal variables.

Sphagnum capillifolium rubellum and *S. papillosum* were the most widespread peat-forming species and M19 the most frequent NVC community. Variation in the main dataset could not be explained successfully by any of the measured environmental variables, either individually or in combination (the models only explaining between 9 and 17% of the variation in composition in the full dataset). In contrast, peat moisture content and pH/litter depth were most influential in the data-subset: species responses to these may be useful when considering the impact of both burning and grazing. It was suggested that "some *Sphagnum* species are more tolerant of dry and low litter conditions,

implicating management practices which cause drying and reduced litter levels, such as burning." Although some *Sphagnum* species were suggested to have potential as indicators of management impact because of their widespread distribution and relative abundance, in particular *S. papillosum* as a positive indicator and *S. capillifolium rubellum* as a negative indicator, though it was suggested that more research is required to validate this.

O'Reilly (2008) also considered the effects of burning on Sphagna in peatlands as part of a wider literature review on Sphagna and reported a paucity of empirical data, as was suggested is the case for the effects of fire on bryophytes in general. Even though fire is widely considered to be detrimental, it was suggested that Burch's (2006) study (above) of a managed burn chronosequence on wet heath in the North York Moors showed Sphagna to be resilient. The long-term study of burning on upland peatland vegetation based at Moor House NNR failed to distinguish different species of Sphagna; other more recent publications present no empirical data for the reported fire response of individual species. Key drivers of change on peatlands following fire that may affect Sphagna indirectly or directly are: a short term increase in pH and nutrient availability (3-5 year duration with a probable long term reduction in P); substrate drying; and volatilisation and redistribution of nutrients in particulate form.

Harris *et al*. (2011b)/Harris 2011 [2+] examined the effect of management burns on plant community composition using a chronosequence from five species-poor *Calluna*-dominated, severely modified, degraded blanket bog sites in the Peak District. Species cover was assessed with respect to elapsed time since burning and vegetation height. Burn patches, of age range 2 to 18 year with a smaller number of unburned (>35 year) patches, were chosen using an age-stratified random sampling method and sampled with random quadrats over three years.

Calluna was the only species to show an increase after burning (and with increasing vegetation height); all others showed an increase immediately after burning and then either decreased or showed a unimodal/skewed response. Most other species were restricted to vegetation <40 cm tall and up to 20-25 year since burning probably as a result of increasing *Calluna* dominance. Species-richness declined (slightly from a low starting level) over time since burning and was lowest in the relatively small number of unburned (≥35 year) stands. The environmental variables analysed explained 15.2% of the variation in species composition. When shared variation was removed, the amount of variation was explained by site (3.6%), production (2.2%), biotic (1.0%) and physical (0.5%) (all significant at $p < 0.0001$) factors. The authors suggested that to maintain the higher species richness in the post burn phase requires rotational burning to avoid *Calluna*-dominance, although with the exception of *Rubus chamaemorus* and some bryophytes, most of these species were acid grassland or heathland species rather than those characteristic blanket bogs.

Palaeoecological studies

Though there are a relatively large number of paleoecological studies from upland peatlands, non-UK studies were sifted out and only two were included from UK where there were clear implications and/or recommendations for current burning management.

Chambers *et al.* (2007) [3+], and in a series of earlier papers and reports, chronicled the palaeoecology of degraded blanket mire in Wales to provide an understanding of various factors in mire degradation and the implications for conservation management. It was noted that many European blanket mires are degraded and contain few Sphagna with more than half exhibiting symptoms of degradation in Wales. Palaeoecological techniques comprising plant macrofossil analysis, charcoal analysis, spheroidal carbonaceous particle (formed from high-temperature combustion of fossil fuels) analysis, pollen analysis, radiocarbon dating and determination of peat humification from two sites in south Wales, one NVC community M25 *Molinia* mire on relatively shallow (<50 cm) peat and the other M18 *Calluna-Eriophorum* blanket mire.

The data collected suggested a major vegetation change which post-dated the start of the industrial revolution. There was evidence for increased burning activity, but as this was not evident in all profiles it was suggested that this was unlikely that fire was the principle or sole agent in vegetation

change. Rather, increased atmospheric input, plus a change in grazing pressure, may have been responsible. The overwhelming dominance of *Molinia* at one site and local dominance of *Calluna* at the other was considered unprecedented. Millennial-scale dominance of Autin's bog-moss *Sphagnum austinii (imbricatum)* characterises the earlier record with its demise and that of round-leaved sundew *Drosera intermedia* took place in historical times. Thus, both sites show floristic impoverishment within the 20th Century, with recent single species dominance. The authors, therefore, suggest that "conservation management to reduce the current pre-eminence of *Molinia* would not run counter to long-established dominance, so in cultural and historical terms can be fully justified. Potential intensive restoration techniques include use of herbicides, mechanised destruction of long-established *Molinia* tussocks, and re-seeding with *Calluna vulgaris* (Anderson *et al.* 2006, above). Our data suggest the best prospects for wider success in South Wales would involve modifying grazing regimes to reduce the prevalence and intensity of sheep grazing, and encourage instead lighter grazing by cattle; reducing burning and atmospheric pollution; and combating gullying to maintain hydrological integrity."

Ellis (2008) [2+] studied the stratigraphy of three cores of entire peat depth and a monolith from Western Rannoch Moor, north-west Scotland. All were from relatively shallow peat, c.1 m deep, on NVC community M17 *Scirpus cespitosus – Eriophorum vaginatum* 'western' blanket bog. Pollen analysis was used to match synchronous stratigraphic horizons between the cores in the same peat system, based on previously identified pollen zones. Five horizons from the monolith were submitted for radio-carbon dating to calculate calibrated years before present (cal. Yr BP). Macrofossils (including macroscopic charcoal) were examined from 1 cm3 samples extracted from the cores at 2 cm intervals. Macrofossils were scored into four contrasting plant groups: fringe-moss *Racomitrium*, *Sphagnum* species, Ericaceae and monocotyledon remains (mainly *Eriophorum* and *Scirpus* spp.). Humic acid was measured from 1 cm^3 peat samples extracted from the cores as 1 or 2 cm intervals. Humic acid is used as a proxy measure rate of decomposition and therefore of surface moisture.

The main findings included: higher levels of macrofossil charcoal were associated with lower levels of surface moisture; higher *Sphagnum* and monocotyledon macrofossil remains associated with increased surface moisture; percentage charcoal was negatively correlated with *Racomitrium* and monocotyledon remains; percentage frequencies of *Ericaceae were* negatively correlated with monocotyledon remains; *Racomitrium*, *Sphagnum* and *Ericaceae were* positively correlated with one another. These results were used to suggest a hypothetical framework for environment-vegetation interactions during millennial scale peatland development. Contrasting plant groups respond to changes in local surface hydrology driven by climatic variation and landscape-scale change (for example, deforestation). Drier conditions, along with more burning are associated with more *Ericaceae and Racomitrium*. Wetter conditions, along with less burning are associated with more monocotyledons and *Sphagnum.* However the physical processes of hydrology and burning may be modified by long-term vegetation interactions, for example, that monocotyledons are negatively associated with *Ericaceous* (through competition) and that *Ericaceous and* the mosses *Racomitrium* and *Sphagnum* are positively associated (through facilitation). Separating the confounding effects of surface hydrology and burning on long-term vegetation development is difficult and further complicated by evidence that species interactions may have modified the vegetation response. The results indicate that while burning has been a long-term feature of the vegetation is did not necessarily uniformly encourage Ericaceous shrubs and preclude *Sphagnum.* Remains of *Erica*ceae were not associated with the charcoal record, but with *Sphagnum* and *Racomitrium*. These positive associations may be attributed to the regenerative growth of Ericaceous shrubs by layering into an intact moss carpet and the sustained growth of mosses around a structural matrix of Ericaceous shrubs. It was suggested that blanket bog vegetation is relatively stable with vegetation dynamics able to modify and buffer the effects of larger scale climate and burning effects. However the loss or change of plant groupings, for example, through pollution or overgrazing, may disrupt this stability.

Modelling

Chapman *et al.* (2009) [2-] developed a spatially explicit model coupling moorland vegetation dynamics and management decisions about sheep grazing and heather burning parameterised for the Peak District National Park (NP), to reflect how real life management decisions can impact on the

dynamics of ecological systems. The model was based on data from 71 management units (MU) of median area 54 ha within the NP, which equates to 40% of the region, from which were derived pragmatic management scenarios submitted to the model. Land units of 100 m x 100 m were defined in terms of the aerial imagery land cover of dwarf shrub, bracken, graminoids, bare peat and rock, modified by: a measure of habitat quality derived from meteorological and physical site data at the same resolution; and simplified inter-specific competitive dynamics. Dwarf shrub was categorised into growth phases depending on time since last burn; degenerate stands were assigned a reduced vigour and expansion in newly burnt stands was constrained. Sheep grazing within each MU was dependent on flock size and the attractiveness of individual land units, determined by availability of preferred forage; the grazing pressure on dwarf shrubs was predicted using the Hill Grazing Management Model. Simulations of the model were initiated by randomising the age of cells within the range 0-30 years and vegetation cover in proportion to habitat quality. Sensitivity analysis was performed on the major parameters not fitted to local data.

The model was used to explore: the effects of current management scenarios (with 10 runs of 500 iterations per comparison) when either summer or winter grazing rate is fixed; burning management continues or is banned; fixed or flexible management; and progressive incremental temperature increases up to the current maximum. Results suggested that maintaining current management in the long-term should reverse historic losses of dwarf shrubs at the expense of graminoids, but ultimately grazing rates must then fall. Enforcing a grazing rate prompts adaptation of management in response to changes in dwarf shrub cover. A progressively warmer climate may favour bracken expansion but prevent bare peat being re-vegetated; burning will decline and dwarf shrubs will respond by a disproportionate representation of degenerate stands.

Systematic review

Stewart *et al.* (2004, 2005) [1++] carried out a ‘systematic review’ on the subject: does burning degrade blanket bog and wet heath in the UK and Ireland. Only eight studies meeting the strict selection criteria were included which reported 11 independent datasets. Of these, seven were of blanket bog vegetation types (all NVC community M19) and four wet heath or degraded blanket bog (M15). Attempts at quantitative analysis of the datasets from these studies were severely constrained by the nature of the data and vote-counting proved the most effective means of data synthesis. The primary measure of effect was favourable condition based on Common Standards Monitoring (CSM) tables. All of the included studies (Adamson 2004 pers. comm. to Stewart *et al.* 2004; Elliott 1953; Forrest & Smith 1975; Hale & Cotton 1988; McFerran 1991; Rawes & Hobbs 1979; and Ross *et al.* 2003) apart from one, Loftus (1994), were also assessed in the present review (see above) based on these or subsequent publications but also drawing on Stewart *et al.* (2004). Though it was not possible to obtain a copy of Loftus (1994), a brief assessment is given above based on Stewart *et al.* (2004), especially the summary given in Appendix 3. It should be noted that five of the 11 datasets are from Moor House NNR, including four apparently from the Hard Hill plots which are described together above supplemented by data from other publications on this ongoing, long-term study. The other sites were in the South Pennines (2), Ireland/Northern Ireland (2) and Northumberland (1 site, 2 datasets). Only the Hard Hill studies involved repeated rotational burning; the others reported on post-burn recovery up to 11 years after the burn, four of which were management burns, four experimental burns and one a wildfire.

The 11 datasets showed a range of changes in floristic composition in response to burning. Overall, the changes tended to: promote the dominance of a few species (four datasets) or switch dominance from ericoids to graminoids (two datasets); increase the quantity of bare ground (four datasets); result in decreased abundance of key species (in CSM assessments) (nine datasets) although two datasets increased the abundance of key species without promoting dominance; and result in minor changes in floristic composition (two datasets). Of these outcomes, the first three were interpreted as resulting in unfavourable condition (as defined by upland CSM tables) and were therefore considered indicators of degradation, whilst the last outcome was considered neutral in terms of favourable condition. When changes were detected that might enhance condition, they were accompanied by negative changes and were therefore considered contradictory with the exception of Elliott (1953). Three datasets suggested that burning degrades blanket bog, whilst five presented contradictory

outcomes and three indicated that burning does not degrade blanket bog. Site comparisons generally suggested that burning does not degrade bog (three datasets) or has contradictory effects (two datasets) whilst randomised controlled trials suggested that burning is contradictory (three datasets) or that blanket bog is degraded by burning (two data sets).

Overall, Stewart *et al.* (2004) concluded that "the weight of available evidence suggests that burning either degrades blanket bog, including wet heath, or is contradictory in effect. If quality of evidence is used to discriminate among studies then the evidence for degradation becomes stronger. However, a degree of circumspection is required given the small sample size, variable timescales of the studies and problems in the interpretation of favourable condition. Pending further research it is suggested that burning on blanket bog and wet heath should normally be avoided if favourable condition is to be achieved or maintained, but there is not a robust evidence-base to support management decisions regarding burning on blanket bog."

Other reviews

The following provides a summary of findings from previous reviews that relate to the effects of burning on upland peatland vegetation that were used in the development of evidence statements or in the text discussing them (see paragraphs 4.18-4.30 in the main text). These draw mostly on the results of studies included in the present review, though in some cases reference is made to other documents not otherwise included (ie not identified in the current review searches or sifted out). In addition, the findings of other evaluated reviews that relate to the sub-question but which are not quoted in the summary and discussion below are also summarised separately in Appendix 2.

Coulson *et al.* (1992) [4+] in a comprehensive review of the management of moorland areas to enhance their nature conservation interest summarised the impacts of burning on blanket bog. They noted that "the growth phases which *Calluna* passes through when growing on dry heath (Watt 1955) do not occur on blanket bog. Instead, due to the constant reburial of stems by *Sphagnum* and the rejuvenation of these stems, an uneven aged stand of *Calluna* is produced, a so called 'steady state' system (Forest 1971; Jones *et al.* 1971; Rawes & Hobbs 1979)." "The general opinion is that blanket bog in comparison to northern heath should be burnt on a longer cycle or not burnt at all (Mcvean & Lockie 1969; Hobbs 1984; Mowforth & Sydes 1989)." They suggested the following reasons for this in relation to vegetation. "Burning on wetter bog areas tends to favour the dominance of monocotyledons over *Calluna* (Sydes & Miller 1988). After the burning of *Calluna*/*Eriophorum* blanket bog, the *Eriophorum* regenerates more quickly than *Calluna*, and it is not until 20 years have passed that the dominance of *Calluna* is re-established (Rawes & Hobbs 1979). Therefore the consequence of burning of blanket bog on a cycle shorter than 20 years means that burning occurs when *Eriophorum* is dominant, hence post fire vegetation contains a more vigorous regrowth of this species compared to *Calluna*; at the time of burning *Calluna* is relatively scarce and can only produce limited regrowth after the fire, and the establishment of *Calluna* seedlings is prevented by the *Eriophorum* (Hobbs 1984). In northern Scotland, on the peatlands of Caithness and Sutherland, the place of *Eriophorum* is taken by *Trichoporum cespitosum*, and in the west of Britain *Molinia caerulea* is co-dominant with *Calluna*. Both *Trichophorum* and *Molinia* become more dominant under more frequent burning regimes (Currall 1981; Lindsay *et al.* 1988)." "There will be a general decline in the species richness of plants on burnt compared to unburnt blanket bog (Hobbs 1984) with *Sphagnum* being especially susceptible to burning. Loss of species such as *Sphagnum* may have important implications for nutrient cycling and erosion." "The use of severe fires on blanket bog often causes the removal of the acrotelm, causing the formation of a surface layer of algal origin which has low permeability (Conway & Millar 1960; McVean & Lockie 1969). The subsequent flow of water over this surface then tends to lead to erosion (Ingham 1983)." The authors concluded that "Northern heath and blanket bog should be managed differently with respect to their burning regimes. It is recommended that blanket bog should not be burnt at all because of the loss of nutrients and plant species which maintain the naturalness of the community."

Shaw *et al.* (1996, 1997) [4+] carried out an extensive literature review on the historical effects of burning and grazing of blanket bog and upland wet heath. The authors reviewed the effects of burning and grazing on NVC blanket mire and wet heath NVC mire community types (M16-20)

drawing particularly on Rodwell (1991) but also a wide range of other sources. They suggested that all of these communities can be impoverished or converted to less species-rich, often graminoid-dominated communities. For example, “many stands of M15 wet heath have been derived from blanket mire communities as a result of climate change, drainage, burning and grazing” and “severe burning can convert M15 into various forms of acid grassland, for example, U4 and U5.” M16 “is protected to some extent where the ground is wet, but burning can be particularly deleterious to the associated flora, with the typical bryophytes only recovering very slowly, and these may be eliminated by frequent fires.” Burning M17 “has a particularly drastic effect on the *Sphagnum* cover, even very wet carpets becoming susceptible to fire-damage in periods of drier weather in spring and summer, and it produces the kind of dominance by (*Trichophorum*) tussocks and *Racomitrium* hummocks characteristic of some tracts of the *Cladonia* sub-community (McVean & Ratcliffe 1962);” “*Eriophorum vaginatum* may become locally dominant after fire and in other cases burning may impose a fairly uniform dominance of (*Trichophorum*) or *Molinia*.” In M19, “species which die back at the end of the growing season (for example, *E. vaginatum* and *Rubus chamaemorus*) may survive winter and early spring fires” and “*E. vaginatum* may dominate for up to two decades after burning (Eddy, Welch & Rawes 1969, Rawes & Hobbs 1979, above).” In M20, “burning and grazing (especially over a long period), are particularly important in the reduction of cover, or even elimination of some of the major sub-shrubs; burning can result in the total destruction of the above-ground parts and sometimes the stools and rhizomes of the ericoids (Gimmingham 1960; Bell & Tallis 1973: cit. Rodwell 1991), stimulating an expansion in the abundance of *E. vaginatum* (Wein 1973; Rawes & Hobbs 1979) and of *Rubus chamaemorus* (Taylor 1971; Taylor & Marks 1971: cit. Rodwell 1991). In some places, M20 may represent just a temporary phase of vegetation following a burn, although repeated fires may maintain the community by continually setting back regeneration of the sub-shrubs.”

The authors refer to evidence suggesting varying responses of *Sphagnum* species to burning, but note that they “have found no studies which specifically address the question of the impact of controlled burning on and subsequent recovery of Sphagna on blanket bog which could help to provide definitive guidance on the appropriateness of occasional winter fires for management.” They concluded that “there needs to be a much greater emphasis on the distinction between ‘wet’ and ‘dry’ upland habitats, as these are often lumped together as ‘moorland’, despite their ecological differences” and that “where carried out without regard for wildlife, regular burning regimes and heavy grazing are likely to be damaging to the wildlife interest of blanket bog and wet heath and can lead to loss of these habitats. However, if carried out sensitively, burning and grazing can be advantageous to some, but necessarily all, species of these habitats.” They noted that appropriate management recommendations depend to a large extent on the objectives but that “the results of our literature review suggest that burning is not usually recommended for management of blanket bog, although there may be a case for its infrequent use in some circumstances.”

Anderson *et al.* (2006) [4+] assessed the potential role of burning as a management tool in the restoration of *Molinia*-dominated upland blanket mire in mid-Wales, and specifically the Elenydd Estate, based on a literature review and fieldwork on the estate. It was concluded that evidence from experimental studies demonstrates that burning without adequate grazing, or over-frequent burning with or without grazing, leads to what can become overwhelming dominance by *Molinia* over large tracts of land. This is a reflection of the ability of *Molinia* to exploit increased nutrient availability following a burn; its inherently high relative growth rate; its ability to sequester nutrients for rapid recycling early in the year; and its basal internodes which are protected from damage by fire. Control of *Molinia* in post-burn vegetation is most effective using less selective grazers such as cattle and ponies, particularly during summer. It was suggested that that burning can be cost-effective as a follow up to herbicide treatment to remove surface litter, thus increasing the likelihood of establishing a wider diversity of plants within the target vegetation.

Appendix 4 Studies and evidence for the effects of burning on fauna

Invertebrates

Nine studies, all but one from the UK (four from England and one from England/Scotland, Scotland, Northern Ireland and Wales), with the remaining study in Germany, provide evidence on the effects of burning of peatlands on invertebrates. In addition, the findings of two reviews that relate to upland peatland invertebrates are also summarised in Appendix 5.

Studies on the aquatic invertebrates of upland peatland watercourses are dealt with under the water quality/flow sub-question (d) and are not repeated here under this sub-question.

Coulson (1988)/Butterfield & Coulson (1985, 1983)/Coulson & Butterfield (1986) [2-] studied the structure and importance of invertebrate communities on 'peatlands and moorlands', and considered the effects of environmental and management changes. Butterfield & Coulson (1985) sampled invertebrates at 42 sites in northern England, 33 of which were on peat. Invertebrates were captured using pitfall trapping and extraction of soil samples. An index of similarity was determined between eight paired peat and mineral soil sites within 400 m of each other. Cluster analysis was used to cluster sites based on the similarity of their species composition. Five main communities were identified including high and lower altitude blanket bog, edge peat and mixed moor (wet/dry bog/heath). Although similar numbers of species were found across the habitats, the mixed moor had the highest diversity in terms of number of species and individuals, although the numbers of individuals caught on the moorland habitats was greater than in lowland mires.

Coulson (1988) further investigated the data including the effects of environmental and management changes. Standing crop showed marked differences between communities. The species of the blanket bog community were typical of those found in sub-arctic regions of Scandinavia and have a northern European distribution. It was suggested that "the mobility of most invertebrates and the relatively small plots which are burnt at any one time raises no major problems for recolonisation for invertebrates. However, it is difficult to separate the direct effects of burning from those associated with the loss of food for invertebrates. Large and extensive burning of a moor, as occurred in the North York Moors in 1976, has had more pronounced effects on the whole ecosystem because of the much larger areas involved, the major effects of the hot fire on the vegetation and burning of peat for many days."

Curtis & Corrigan (1990) [2-] investigated the relationship between the diversity and composition of spider communities and land management/vegetation of blanket bog and wet heath at six sites under varying management, including burning, on Islay, Scotland. Spiders were collected using pitfall traps (mean 26/site) over one year (resulting in capture of c.9,200 individuals).

Differences in spider community assemblages (species diversity/richness and species/species groups were attributed to changes in the vegetation composition and structure/microtopography as a result, directly or indirectly, of burning and grazing management practices. Shorter swards resulting from burning and grazing were associated with relatively high species richness, but low species diversity reflecting increased numbers of surface-active *Lycosids* but declines in numbers of web-spinning *Linyphids*.

Usher (1992) [2-] examined the diversity of ground beetle (family *Carabidae*) and spider communities at 52 'sites' on three North York Moors grouse moors (peat areas not specifically separated), 25 described as 'in heather', 17 'burnt' and 12 'cut'). Ground beetles and spiders were collected using a mixture of pitfall traps, water traps and suction sampling.

Analysis of the data using ordination and classification methods suggested that species assemblages from burnt sites differed from unburnt sites, for both beetles and spiders. For example ordination of the data for the 30 most common species of ground beetle resulted in the unburnt and uncut sites being concentrated along one axis, the other axis probably being associated with altitude. Species assemblages were primarily considered to be influenced by the growth-phase of heather. Species of open ground, such as *Xysticus sabulosus* and *Gnaphosa leporine* (spiders) or *Calathus erratus* and *Nebria salina* (beetles), were only trapped infrequently in building and mature phase heather. In contrast spider species in the genera *Lepthyphantes* and *Walckenaeria* tended to be more commonly caught in the 'heather' than in the 'burnt'/'cut' areas. Some of the nationally rarer species were associated with open conditions of recently burnt heathland. It was concluded that the mosaic of heather growth-phases resulting from burning/cutting appears to be responsible for maintaining a diversity of faunal habitats and hence maintaining the large number of arthropod species found in upland heathlands.

Holmes *et al.* (1993) [2-] carried out a large-scale survey of ground beetles (*Carabidae*) in a total of 300 trapping stations in 118 peatland sites throughout Wales. Factors influencing the distribution of carabid beetles were investigated using ordination. Burning (categorised as recently burnt, burnt but not recently and not known to have been burnt) was one of the variables recorded.

The most important factors identified were nutrient status and saturation of the substrate (which may be affected by burning), altitude and livestock grazing. Burning was also considered a significant factor but could not be investigated thoroughly because of insufficient data on timing, frequency and severity of burns.

McFerran *et al.* (1995) [2+] studied changes in the composition and structure of *Calluna*-dominated wet heath and the effects on ground beetle and spider assemblages in response to burning and grazing in site comparisons (described further under sub-question a) at Ballycastle, in County Antrim, Northern Ireland. The aim was to identify patterns of invasion and possible successional sequences. Three replicate pitfall traps were installed in each plot and the preserved contents collected at 4-6 week intervals over an 18 month period in 1989-90 with data combined to produce a single observation. Abundance data were loge transformed and subject to ANOVA. Assemblages derived from presence/absence data for each taxon were analysed by multivariate ordination and classification. The relationship between ordination axis scores and botanical variables derived from point quadrat data were explored by rank correlation.

The results suggest that burning has more effect on the assemblage of invertebrates than a short period without grazing and that the patterns of post-burn invasion over time are different for ground beetles and spiders. The most distinctive assemblage of ground beetles was found after the more recent1988 burn, which had the highest catches of *Carabis nitens* and *Nebria salina* ($p < 0.05$ & 0.001, respectively) but the lowest of *Pterostichus niger* and *Leistus rufescens* ($p < 0.001$ and 0.01, respectively). Most *Carabis glabratulus* were recorded from unburnt areas ($p < 0.001$). In contrast, the greatest numbers of spiders were trapped after the older 1982 burn ($p < 0.01$), where five species were at their most abundant (p-values of 0.01 or 0.001); three species were least prevalent among newly burnt vegetation ($p < 0.01$ and < 0.05 for 1 species). Successional trends and the impact of grazing were discriminated poorly by both ordination and classification although 1988 plots showed some separation on axis 2 for ground beetles, while the presence of *Carabis nitens* distinguished these samples at the first division of the classification. No relationships between axis scores and key botanical variables were elucidated. Nevertheless, the authors argued that there was evidence that distinct assemblages and/or individual species can be related to the changing composition and structure of *Calluna*-dominated heath. The list of ground beetle species recorded at Ballycastle were compared to a previously published classification derived for 42 peat sites in northern England (Butterfield & Coulson 1985 [2-]) and corresponded most closely to type 1, typical of dry heath-like moorland of an appropriate altitudinal range and peat depth.

Dennis & Eales (1997, 1999) [2+] investigated the effects of habitat quality, isolation and patch area on occupancy of the large heath butterfly *Coenonympha tullia* in Northumberland. The county has around 72% of the known occupied sites for this nationally declining species, mostly on blanket,

intermediate and raised bogs. Dennis & Eales (1997) identified 166 sites for survey; 90 identified from previous survey and 76 identified as potential sites from habitat maps. For each site presence/absence of the butterfly, isolation from nearest occupied patch, patch area and measures of habitat quality were recorded.

The large heath was found at 76% of sites surveyed. All sites occupied are between 140 and 410 m OD and have a mean size of 22 ha (range 1-170 ha). They range in isolation from 10 m to 30 km. Habitat quality, patch size and isolation account for 61% of variation in occupancy. Though bogs are the habitat most likely to be occupied, they were also found in wet and dry heath, acidic and marshy grassland. Sites where the butterfly was not found had a greater proportion of heath than bog. Most favoured habitats were those with dense, vigorous growth of *Eriophorum vaginatum* and *Erica* tetralix on sheltered mires. Frequent burning was considered likely to render sites unsuitable, but occasional burning may be beneficial in reversing succession. Larger sites and those closest to other occupied sites were most likely to be occupied. Dennis & Eales (1999) built on the previous study by surveying more sites and trying to predict occupancy from the habitat data, site size and site isolation data. Large heath was found at higher altitude sites (mean 321 m) and more frequently on heath than in previous survey.

Eyre *et al.* (2003) [2+] investigated the distribution of ground beetles, rove beetles, spiders and plant bugs in a five-year study on a grouse moor in the Langholm-Newcastleton SSSI in SW Scotland. Samples were collected by pitfall trapping and suction sampling at 59 sites in four habitat types including 'wet moor' and *Molinia* moor (and dry heath and grassy streamsides) and subject to four management regimes: unmanaged, sheep grazing, herbicide/burning and burning/cutting. Data were analysed to investigate relationships between assemblages and environmental variables.

Ground beetle assemblages as defined by classification were related to habitat structure rather than vegetation type or management except that one group was predominantly found in bare areas created by *Molinia* control. Management of *Calluna* moor created habitat diversity for ground and rove beetles. Thirty-nine rare or scarce species were recorded for which the most important habitat was streamsides. Some rare species were restricted to sites managed by burning and cutting and others were confined to unmanaged wet moor. *Molinia*-dominated areas were of poorer quality than *Calluna* habitats.

Stone (2006) [2-] investigated vegetation and invertebrate communities on blanket bog in the Peak District. Data were collected from 42 sample areas on three moors managed by rotational burning in *Calluna*-dominated, modified blanket bog in stands characterised by *Calluna* in each of the four stages of the growth cycle spanning one to over 40 years after burning. Stands in each of the four stages were used to represent age classes post rotational burning.

Cover of *Calluna* increased to a maximum in the mature phase then declined. There was a negative correlation between grass abundance and *Calluna* cover. Bilberry occurred in young stands but was generally absent from mature and degenerate stands and cross-leaved heath occurred only where there were gaps in the heather canopy in pioneer and degenerate phases. Invertebrates were collected through pitfall trapping between May and July. 2,507 individual invertebrates were taken including 398 carabid beetles of 25 species. Other taxa collected in numbers were: Arachnids, Opiliones, Collembola, Acari, ants and Hemiptera. Catch per unit effort was calculated to elucidate differences in invertebrate assemblages between *Calluna* growth phases. Overall species richness did not vary between phases. Among non-coleopterans only Opiliones showed a difference in abundance being most numerous in the building phase. Diversity of carabids was highest in pioneer stands and at a minimum in mature stands. Diversity of showed no consistent correlation with environmental variables, with significant correlations with moss cover on one moor with structural complexity of vegetation on another and no correlation with any variable on the third. Species assemblages varied with pioneer phase stands characterised by a group of species that are regarded as mobile and typically found widely in open habitats. These species are replaced as the *Calluna* growth cycle progresses and there is not a return to the community of open habitats as the heather canopy degenerates. A range of heather stand ages is therefore required to maintain maximum carabid diversity at a landscape scale.

Hochkirch & Frauke (2007) [2++] investigated the effects of burning on grasshopper (Orthoptera) species assemblages on degraded raised mires in NW Germany. In these areas burning is used to control vegetation and maintain open structure for bird species. Planned fires are set in winter when the ground is wet or frozen but there are also uncontrolled wildfires in other seasons. Four study areas were sampled: two where there had been winter controlled burning and two where there had been spring wildfires. Orthoptera were sampled along 50 m transects in burned areas and in parallel 50 m transects in adjacent unburned stands. Vegetation composition and structure were also sampled. Data were analysed by ANOVA and NMDS Ordination.

Orthoptera assemblages differed more between mires than between burning treatments, perhaps reflecting known differences in vegetation, size, degree of degradation and size between moors. There were no significant differences between the effect of planned burns and wildfires. Burning did not decrease the abundance of Orthoptera and had neutral effects on most species. One species of conservation concern responded positively to burning. Burning may affect the phenology of Orthoptera through creating open conditions in which temperatures are higher so affecting rates of development. Winter fires may have little effect because Orthopterans are inactive and survive as eggs in the soil or in unburned 'refuges' within burn patches. Vegetation structure is the main determinant of Orthopteran species assemblage.

Reptiles

No studies were found that related specifically to the effects of burning upland peatlands on reptiles, although information from a more general review of burning impacts is presented in Appendix 3.

Breeding birds

Nine studies provide evidence for the direct or indirect (through changes in vegetation/habitat composition and structure) effects of burning on moorland breeding birds, though many do not differentiate peatlands from other moorland habitats. All are UK studies including three from, or differentiating data from, England. Most are large-scale correlation studies between breeding bird numbers/densities and habitat and other environmental/management variables, often including burning or gamekeepering intensity. In addition, the findings of the main reviews that relate to breeding birds are also summarised separately below or, when they are not quoted in the summary and discussion below, in Appendix 3.

Picozzi (1968) [2+] examined the influence of patterns of burning and base-status of underlying geology on red grouse bags in a correlation on 26 grouse moors in Scotland, although the geology and soil types of the sites are not given. The study further investigated previous suggestions that chemical composition of *Calluna* and hence numbers of grouse are affected by the underlying rock (Miller *et al.* 1966). Data were collected on moors for which grouse bag records were available for ten years prior to 1966. *Calluna* growth stage, mean height and *Calluna* cover were recorded every c.60 m on transects across moors, plus the number of recent burns and the area of even-aged *Calluna* stands (by size class) crossed. An index of base status (based on equivalent weights of Ca and Mg present) for each moor was derived from percentage surface area underlain by four classes of rock type grouped by known chemical composition.

Grouse bags were positively related to number of recent fires and base-status. It was suggested that a large number of fires indicates that burn parch sizes were small. There was a suggestion of a weak, but not significant, relationship with area of burning. These results were supported by additional experimental data from a site in Kincardineshire that also showed that grouse bags significantly increased following burning in small patches. The data support the practice of burning in small strips to maximise grouse productivity. Burning on a rotation of 10-12 years with many small fires was recommended (but the vegetation types to which this applies were not defined).

Haworth & Thompson (1990) [2+] researched factors associated with the breeding distribution of moorland birds in a correlation study in the South Pennines. The size of each shooting estate and number and weekly work patterns of gamekeepers were identified. Gamekeepering intensity was

quantified as number of keeper hours per week divided by the number of 0.25 km squares in each estate. Breeding birds were surveyed in 1982 and 1983. Discriminant function analysis was used to compare 0.25 km squares with and without each breeding bird species with a number of variables (including gamekeepering intensity). Although the area includes extensive areas of plateau blanket bog, these were not differentiated in the analysis. No distinction was made between burning and predator control activities by gamekeepers.

Keepering was associated with the presence of three wader species, but the significance was weak compared with topography. Areas subject to intensive keepering were favoured by golden plover, redshank and curlew. Short-eared owl and merlin showed a net preference for intensively keepered moor.

Smith *et al.* (2001) [2+] examined grouse abundance and habitat in 69 1-km squares in upland Britain; meadow pipit *Anthus pratensis* abundance was studied on 36 of these ('among-moors study'). In addition similar data were collected on 73 sites of 25 ha within the Langholm estate in SW Scotland ('within-estate study'). Although the incidence of *Sphagnum* was recorded as one of the habitat variables, no detailed habitat data were provided and it is not clear what proportion of the sites were on peat.

Meadow pipit *Anthus pratensis* abundance declined with increasing incidence of muirburn ($p = 0.004$ within-estate; $p = 0.001$ among-moor), heather ($p = 0.05$ within-estate; $p = 0.008$ among-moor), and *Sphagnum* ($p = 0.03$ among-moor only) suggesting that sites with less burning had more meadow pipits. Muirburn explained 19% of the variation in pipit numbers (among-moor dataset). Increasing amounts of muirburn were associated with more *Calluna* ($p < 0.001$) and less grass at Langholm ($p < 0.001$), but the amount of muirburn was not related to the presence of *Calluna* ($p = 0.09$) or to the amount of grass ($p = 0.93$) in the among-moor dataset. It was noted that a previous study had demonstrated that meadow pipit abundance appears to influence the settling densities of hen harriers (Redpath & Thirgood 1999) and therefore the extent of burning is predicted to influence hen harrier densities. Bird species diversity (as determined by Simpson's index for among-moor data) increased on moors with more muirburn (explaining 9% of the variation, $p = 0.002$). However, there was no relationship between species number and muirburn. In general, more bird species were present on drier moors with less *Calluna* and lower habitat patchiness.

Tharme *et al.* (2001) [2+] assessed the effect of management for red grouse shooting on the population density of 11 species of breeding birds in a correlation study on *Calluna*-dominated moorland. Breeding birds, vegetation and moorland management (primarily predator control and muirburning) were surveyed in 320 1-km squares on 122 estates in upland areas of eastern Scotland and northern England. The main habitat type recorded was 'heath' (c.76%) with <10% 'bog', although heath probably included areas on deep peat. However it was noted that 32% of the 1-km squares on grouse moors was on peat (and 22% on other moors). The extent of burning was recorded by estimating the cover in each quadrant of the 1-km square of three different burn types: burned within 1 year of the survey, burned at least 1 year prior to survey and with no *Calluna* regeneration, burned at least 1 year prior to the survey and with evidence of moderate to good regeneration.

Regression models were used to relate bird density to vegetation, topography, climate and soil type. After adjusting for significant effects of these habitat variables, there was evidence of a positive effect of heather burning on the density of red grouse ($p < 0.001$) and golden plover ($p < 0.05$) and a negative effect on meadow pipit ($p < 0.05$). It was suggested that the analyses may have failed to detect effects of burning on other species because the measures used did not take the size and arrangement of burned patches into account. The selection of *Calluna*-dominated squares resulted in the grouse moors having relatively similar vegetation to other moors. However, grouse moors had less grass/bracken, long vegetation and more flush/grass, and medium and long height heath (all $p < 0.05$ or less). Whether these differences are as a result of burning is not clear; it is possible that moors with these characteristics are more suitable for grouse shooting.

Tucker (2003) [2+] also carried out a preliminary analysis of egg-laying dates of selected upland species (but including lowland records) in relation to the legal period for spring burning (up to 15 April

in the uplands in England) using then unpublished national data from the BTO's Nest Record Scheme. It was concluded that "burning could jeopardise the nests and eggs of a significant proportion (earliest date and date for given percentile given in square brackets for UK unless otherwise stated) of golden plover [28 February, 25% by 17 April], snipe [14 March, 25% by 17 April], lapwing [16 March, 50% by 12 April in England] and redshank [1 April, 10% by 18 April]. Some clutches of hen harrier, skylark *Alauda arvensis*, meadow pipit, merlin, curlew and ring ouzel *Turdus torquatus* may also be at risk." It was also noted that "the overall impacts of such nest losses on breeding success and long-term populations in unclear, but these could be exacerbated by climate change which appears to be resulting in early breeding in many birds. Evidence from the Nest Record Scheme has already indicated that breeding has advanced by at least a week as a result of climate change (Crick *et al.* 1997, Crick & Sparks 1999)."

Tucker (2003) also included a comprehensive literature review of the impacts of heather and grassland burning in the English uplands on biodiversity, soils and hydrology (see description under sub-question a) predicted the likely indirect impacts of burning management on some key moorland birds based on a literature review. His interpretation was that burning may be detrimental for hen harrier, merlin and short-eared owl if some old heath is not retained for nesting, but that it may be beneficial for black grouse *Tetrao tetrix*, golden plover, skylark and twite *Carduelis cannabina*.

Moss *et al.* (2005)/Crick *et al.* (2006)/Newson *et al.* (2007) [2++] studied the timing of breeding of birds on moorland in England, Scotland and Wales based on data on species which are more-or-less restricted to the uplands and data from other species' records from within the three Countryside Survey 2000 upland Environmental Zones (Haines-Young *et al.* 2000). They used two large national datasets: the Nest Record Scheme (NRS, 4,284 Nest Record Card (NRC) records) and ringing of nestlings (26,043 records); and two smaller data sets: the GWCT's long-term data set on Scottish red grouse (318 nests); and the RSPB's hen harrier nest data (1,235 nests although only 17 were from England). These data include but do not differentiate nesting on upland peatlands from other moorland habitats. Newson *et al.* (2007) updated the findings in relation to Wales.

The study presents the dates of laying of the first egg in a sample of nests for 23 'priority' moorland species and in summary form for a further 12 'subsidiary upland species'. There was generally good agreement between data from the NRS and ringing. Of the priority species, only nine (39%) had 10% or more (all but one 20% or more) of first eggs laid in nest attempts in England by 15 April (the end of the legal burning season). Among these were some potentially vulnerable ground/vegetation-nesting species: lapwing, golden plover, redshank, snipe, short-eared owl and stonechat *Saxicola rubetra*. A further two species had more than 5% of first nest attempts by 15 April in England based on NRS data: hen harrier and skylark. Of the subsidiary species, five out of 12 had 5% or more of first eggs laid by 15 April, though none were ground/vegetation-nesters. Additional GWCT Scottish red grouse data confirm the relatively late breeding of this species with few first eggs predicted to be laid by 15 April, though there was variation between years and some evidence of later laying with advancement of a week between 1992 and 2003. Laying dates were generally later in Scotland than in England and Wales. Analysis of trends in breeding performance showed that the commonest pattern was for declines in clutch (6 of 13 species) and brood size (4 out of 9 species) through the breeding season but no significant trends in other variables. Trends in overall productivity per nest attempt could only be a few species: wheatear and whinchat showed a decline and twite an increase followed by a decline. It was suggested that losses of nest during April "could have substantial impacts on the productivity" of some species at risk from burning including: wheatear and less so for hen harrier and stonechat, even if the birds could relay.

Regression analysis showed that a number of species nesting earlier than at the start of the data collection (1939 for NRS) perhaps "reflecting generally earlier breeding seasons attributed in other studies to global warming", although a smaller number of species were also laying later. Five species showed earlier laying based on NRCs and seven based on ringed nestlings. Amongst those potentially at risk from burning, these included hen harrier and lapwing becoming earlier by more than one day per year (from NRC data) and ring ouzel and twite *Carduelisflavirostris* by more than 0.3 days per year (from ringing nestlings). A literature review was also carried out on pre-nesting periods, including that dates that migrant or partially migrant species return to moorland breeding

grounds, as an indication of when they would potentially be liable to disturbance from burning operations. Resident species, including red and black grouse, and several waders including golden plover, lapwing and curlew, might be affected particularly in March.

The authors noted that whilst "these results can be used to indicate the vulnerability of moorland birds to burning, they do not show what proportion of nests is actually affected. This true vulnerability may depend on aspects such as the choice of nest sites in relation to types of heather that are burnt, which may in turn vary depending on the objectives of burning. For example, golden plover tend not to nest in stands of mature heather that are ready for burning as part of grouse moor management (though they nest in shorter and fragmented heaths, for example, Ratcliffe 1976), but may be affected if a fire spreads into other more suitable habitats. They may also be affected by 'swaling' ... on grass moorland. A number of additional factors must be taken into account to assess that risk: the proportion of suitable habitat subject to management through burning; the frequency with which a managed moor is burned; and the effect of burning operations on the species' nesting attempt." They went on to suggest potential approaches to creating a vulnerability index, but noted a relative lack of data on some key aspects and did not develop it further. However, they concluded that moving the legal burning cut-off date back to 31 March "would remove many species from significant risk." Following reviews of the Burning Regulations and Code, this subsequently happened in Wales (WAG 2008) but not in England (Defra 2007).

Daplyn & Ewald (2006) [2-] analysed data from two moorland breeding bird censuses conducted within the Peak District in 1990 and 2004 to determine the effect of gamekeepering and managed burning for red grouse on moorland breeding birds. Additional data on raptor numbers were available for 2004. A GIS system and generalised linear modelling was used to relate the density of upland bird species in moorland management units in 1990, 2004 and the change between the two census dates to: the proportion of four broad habitat types determined from data for the ESA published in 1988; the prevalence of burning as determined from aerial photographs from 2001; the presence or otherwise of keepers associated with red grouse management; and the density of keepers employed on estates with shooting.

Moorland units managed for grouse experienced more burning and contained more heath and mire than unmanaged units that contained more grass habitat. The densities of curlew, lapwing and ring ouzel were higher on moors where there was more burning whereas twite, skylark and wheatear declined as the proportion of land burnt increased.

Pierce-Higgins & Grant (2006) [2+] investigated relationships between moorland breeding bird abundance and vegetation composition and structure in a large correlation study. Data were collected on bird abundance using standard survey methodologies, vegetation attributes and non-vegetation factors. Neither peatland habitats or burning were specifically considered although they relate to some of the vegetation composition and structure variables. Data were collected from 85 2 km^2 plots in southern Scotland and northern England selected initially as a random sample stratified for *Calluna* cover. Access permissions were denied to some plots and these were replaced by plots chosen non-randomly for ease of access. Sufficient data to allow discussion of relevant factors were collected for nine species. Multivariate analyses were conducted with effects of non-vegetation factors accounted for before vegetation factors were used to explain the remaining variation in the bird data. Autocorrelation effects between vegetation and non-vegetation factors were considered.

Red grouse and stonechat were associated with heather but both favoured some heterogeneity in cover and the latter was associated with tall vegetation. Snipe and curlew were most abundant where vegetation structure was heterogeneous. Skylark and golden plover were associated with short vegetation, especially short grass and short dwarf-shrub cover. Wader species were positively associated with plant species indicative of wet conditions. Whinchat were associated with dense vegetation especially bracken. No strong vegetation effects were noted for wheatear. Meadow pipit was not affected by structure of vegetation but favoured grass-heather mixes. Heather losses (for example, due to overgrazing) were considered likely to have affected only habitat availability for red grouse and stonechat. No strong links between heather cover and breeding waders were found. It was suggested that changes in vegetation structure and overall heterogeneity (which could result

from burning) may have greater effects on several species rather than changes in vegetation composition. It was suggested that species associated with short vegetation could be affected by future changes in livestock numbers.

Amar *et al.* (2009) [2++] explored whether changes in the abundance of five wader species (Golden plover, Lapwing, Dunlin, Curlew and Snipe) in the uplands correlated with key hypotheses proposed for their declines. The research used data from the 'repeat upland bird surveys' (RUBS) examined at two spatial scales (region and plot) to ascertain whether population changes correlated with arrange of environmental/management variables including broad habitat (including bog as a category) and burning (using an intensity index derived for 10-km squares from satellite imagery). Data from 142 plots was used in the analyses distributed across ten survey regions in Scotland, England and Wales. The analyses attempted to assess empirically the balance of evidence for different broad-scale land-use and habitat-related drivers of upland wader population declines. While the analysis tests for correlation rather than causality, it provides pointers to the respective roles in driving wader declines.

At the plot-scale, curlew and lapwing declined more on *Calluna*-dominated plots than bog plots. Golden plover showed greater declines at the plot-scale where burning was more intensive in contrast to predicted results. However, lapwing showed near-significant trends of greater declines at the plot-scale in less intensively burnt plots and greater declines in regions with less burning at the regional-scale.

Fletcher *et al.* (2010)/GWCT (2010) [1+] investigated the impact of predation on upland breeding birds in an eight-year field experiment in which the abundance of legally controllable predators was manipulated whilst maintaining consistent habitat conditions. This was carried out at four moorland/marginal farmland study sites in Northumberland. Numbers of seven ground nesting bird species were monitored and breeding success of five of them. Sites were paired, one with predator control and the other none, and control was switched halfway through.

Control was successful in decreasing the abundance of fox *Vulpes vulpes* (-43%) and carrion crow *Corvus corone* (-78%). These reductions lead to a threefold increase in breeding success of lapwing, golden plover, curlew, red grouse and meadow pipit. This resulted in a subsequent increase in breeding numbers of these species apart from meadow pipit. As would be expected, this suggests that some of the increases seen in some species on grouse moors in other studies reflect predator control rather than just burning.

Reviews on breeding birds

The following provides a summary of findings from previous reviews that relate to the effects of burning on upland peatland breeding birds that were used in the development of evidence statements or in the text discussing them in paragraphs 4.18-4.30. These draw mostly on the results of studies included in the present review, though in some cases reference is made to other documents not otherwise included (ie not identified in the current review searches or sifted out). In addition, the findings of other evaluated reviews that relate to the sub-question are also summarised separately in Appendix 2.

Ratcliffe (1990) [4+] in a book on bird life of mountain and upland, summarised the dates for the average onset of laying of breeding birds by which at least 10% of the total breeding population has begun to lay, though the source of these data is apparently not identified. This suggested that the onset of laying was before 15 April (the end of the legal burning season) for seven species, although only two, lapwing and stonechat were ground- or vegetation-nesters potentially at risk from burning. The dates given by Ratcliffe (1990) are later than the corresponding figures for 10% of first nest attempts from the more comprehensive and recent national Nest Record Card and ringing data analysed by Moss *et al.* (2005).

Glaves *et al.* (2005) [4+] in a more general review of the effects of burning on biodiversity, soils and hydrology, reviewed the impacts of burning on birds including data on the timing of breeding of

moorland birds from Ratcliffe (1990), Tucker (2003) and, in particular, Moss *et al.* (2005) in relation to the 15 April end-date of the legal burning season in the uplands in England. Of the nine upland 'priority species' that had 10% or more of first eggs laid in first nest attempts in England by 15 April were six potentially vulnerable ground/vegetation-nesting species (% before 15 April in brackets from NRS): lapwing (56%), golden plover (24%), redshank (18%), snipe (39%), short-eared owl (20%) and stonechat (26%), with a further two species with more than 5% of first nest attempts by 15 April: hen harrier and skylark (both 6%). After reviewing available data on factors that might affect vulnerability to burning, the authors suggested that the proportion of a species' first nest attempts on moorland that might be lost to burning is likely to be no more than 1-2%. Thus, it was concluded that "based on the information currently available, a very small proportion of most bird species are at a risk of losing their nests and eggs during controlled burning. Further, any clutches lost, are likely to be replaced ... so reducing the impact of loss on breeding success. The Panel concludes that the population dynamics and status of upland bird species are unlikely to be affected by continued legal burning in the uplands up to 15 April."

Grant *et al.* (2012) [4+] carried out a detailed review of the costs and benefits of grouse moor management to biodiversity and aspects of the wider environment including the effects on moorland birds. It was noted that burning may affect moorland birds in two main ways: "by direct destruction of nests and through modification of habitat (including the structural characteristics and the availability and condition of both plant and invertebrate food sources." On the former, it was noted that "the available data suggest limited overlap only between the date on which generally has to cease and the time of clutch initiation of many ground nesting birds on moorland (Glaves *et al.* 2005, Moss *et al.* 2005, both see above). Amongst the ground nesting species that show greatest overlap in this respect, several select nest-sites in short vegetation (notably golden plover, lapwing and curlew – Whittingham *et al.* 2002, Robson 1998), where burning is unlikely to occur in grouse moors, but others select nest-sites in tall *Calluna*, where muirburn is likely. Short-eared owls and stonechats may be amongst the most vulnerable species in this respect, given that a substantial proportion of nesting attempts are initiated before 15 April ... and that both species nest in tall *Calluna* (Roberts & Bowman 1986; Moss *et al.* 2005, see above; Pearce-Higgins & Grant 2006, see above)."

Mammals

Miles (1971) [2+] investigated the effect of management burning of a mosaic of *Trichophorum-Eriophorum* blanket bog with abundant *Molinia* and *Molinia*-dominated grassland on grazing utilisation by red deer on the island of Rhum, Scotland (described further under sub-question a).

Grazing was significantly greater in burned than unburned *Molinia*-dominated areas in the spring and summer following burning. Grazing rates declined again in both the two years after burning. Burning reduced litter by 80% on grassland and 85% on bog but within two years the quantity of litter had increased to half that on unburned areas. New growth of *Molinia* on burned ground was attractive to grazing deer and burning therefore increased grazing value but such value was evident only in the year of burning and therefore increases the need to burn to maintain grazing value.

Appendix 5 Other evaluated reviews: descriptions and findings for the effects of burning on vegetation composition, structure and function and fauna

The following provides a summary of findings from the main previous reviews that relate to the effects of burning on upland peatland flora and fauna that are not included in the main body of the text.

Floristic composition, structure and function

Mowforth & Sydes (1989) [4-] in a literature review on moorland management noted that "a 10-year burning cycle will eventually eliminate heather on blanket bog, where productivity is reduced by the impoverished substrate (Hobbs 1984). Hobbs working at Moor House NNR in the north of England, records that *Eriophorum vaginatum* comes to dominate under a ten-year cycle. Heather restoration may take between 11-17 years after burning on blanket bog (Rawes & Hobbs 1979) and heather only regains its full dominance after 20 years (Taylor & Marks 1971; Hobbs 1984)." "Heather on blanket bog does not exhibit the cycle that has been identified on dry heather moorland and burning may not be required to provide rejuvenation of the bushes (McVean & Lockie 1969; Gimmingham 1971; Rawes & Hobbs 1979; Hobbs 1984). Heather stems buried by the growth of *Sphagnum* and other pleurocarpus mosses produce adventitious roots and these continue the growth of the heather stem, so that stems above the moss layer are comparatively young (Forrest 1971; Hobbs 1984; Hobbs & Gimmingham 1987)." They therefore concluded that "a 20-year burning regime is the recommended minimum to maintain heather on blanket bog."

Rowell (1990, 1988) [4-] in a review of the management of peatlands for conservation and the associated Peatland Management Handbook listed a number of blanket bog and wet heath NVC communities for which burning, often in combination with grazing, is considered a major factor in their development: NVC communities M15, M16, M19, M20, and M25. M17 and M18 were also listed as communities that are easily damaged by grazing and "particularly affected by fire owing to a drastic effect on *Sphagnum*." It was noted that "burning of bog vegetation is recognised as causing shifts from *Sphagnum* richness to heather dominance and, eventually, to a distinctly heathy community" and that "burning can be recommended as a management tool only in a few peatland situations" such as an initial restoration treatment of cutover and dried out mire and restoration of reed and sedge (Cladium) beds or where "there is clear evidence that a community has developed under, and is only maintainable by, burning." "Burning of blanket bog should be avoided, and should not be used in NVC communities M17 or M18, in any other *Sphagnum* -rich community, and particularly not in any bog system with pools and ridges."

Thompson *et al.* (1995b) [4+] gave a schematic diagram of the changes in moorland vegetation typically occurring in response to grazing and burning. This suggested that a combination of grazing and burning is associated with the conversion of M18 and M19 blanket mires into generally less species-rich Eriophorum-dominated M20 blanket mire. Similarly, grazing and burning, coupled with drainage, may lead to shifts from M17 or M19 blanket mires to M15 wet heath.

Tucker (2003, 2004a, b) [4+] carried out a comprehensive literature review on the impacts of heather and grassland burning in the English uplands on biodiversity, soils and hydrology. It was noted that there are two key ecological differences between bogs and heaths that affect their response to fire.

Firstly, the typical *Calluna* growth cycles seen in heaths do not normally occur on bogs unless they are degraded because layering occurs as the stems are buried by growing *Sphagnum* and as a result the age of above ground stems is uneven and comparatively low and degeneracy does not occur. Instead the bog achieves a 'steady state'. Secondly, the growth and productivity of *Calluna* on bogs are much lower than on heaths. It was concluded that "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. There is also a high risk of erosion following fires on bogs which may lead to further vegetation impacts and significant losses of sequestrated carbon."

Jones (2005) [4+] reviewed the effects of burning on blanket bog to inform a review of CCW's policy on burning which led to a position of no burning except in limited circumstances (Sherry 2005). "Burning is a widely used management technique in upland regions of the UK. Its most commonly cited agricultural benefits on blanket bog are (i) that it rejuvenates plants such as *Calluna* which otherwise become unpalatable with age, (ii) encourages new growth of graminoids such as *Molinia* and *Eriophorum vaginatum* which are favoured by sheep in the spring, and (iii) removes accumulated litter (Lance, 1983). Fire is also employed to create a mosaic of multi-aged *Calluna* stands to increase the carrying capacity for red grouse *Lagopus Lagopus scoticus*: young shoots in recently burned areas provide food for adult birds, while taller older stands provide nesting sites and cover for chicks. These reported benefits of fire as a management tool on blanket mire contrasts sharply with a substantial body of opinion and published scientific evidence which maintains that even controlled burning is best avoided on blanket mires (Stewart *et al.* 2004). Most generic habitat management guidance and environmental land management schemes adopt a very cautionary or even preventative stance on the issue of burning (for example, Backshall *et al.* 2001; CCW, 1999; Mowforth & Sydes, 1989; Rowell, 1988; SWT, 1995)." Much of the Welsh blanket mire resource is likely to have been subject to anthropogenic burning at some time or other. The remains of charcoal occur widely in Welsh blanket peat profiles (for example, Bostock, 1980, Chambers *et al.* 2000) and fire is regarded as having been an important causal component of blanket mire degradation (Yeo, 1997).

The author stated that "the overwhelming weight of evidence is that burning constitutes a detrimental activity for blanket bog vegetation referable to the NVC types M1-3, 17-20 and species-poor M25 on deep peat. The agricultural argument for its use is weak and often contradictory and the overall conclusion of Shaw *et al.* (1996) was that "there is no evidence that burning will increase the grazing potential for sheep". This being the case, the presumption should be against the use of fire on vegetation referable to blanket mire and on bare peat, irrespective of peat depth. The latter point is important because although 0.5 m is widely used as the threshold peat depth for blanket mire, thinner peats inevitably occur in many contexts – for example at the edge of the peat massif, in areas modified by past peat cutting, and where peat formation only began relatively recently (for example, Bostock, 1980). There is no agreed minimum depth of peat which can support blanket bog vegetation."

It was noted that there are at least two situations which present a less clear-cut suite of issues: (i) where vegetation types not readily referable to those listed above occur on deep peat, and (ii) cases in which vegetation which may or may not be a modified derivative of more typical blanket bog occurs on shallow peats. Both situations occur widely across the Welsh uplands. For example, vegetation with a high cover of *Calluna* and poor representation of Sphagna occurs on deep peat at a number of sites, while a range of vegetation types occurs on shallow peats. Although opinions on this differ within the UK, there is a strong case for considering the former type as peatland habitat. Part of the reason for this is that a continuous gradation of vegetation can be found linking such vegetation to forms classically associated with active blanket bog – in which case where does one draw the line? Furthermore, vegetation which we would nowadays regard as strongly modified (ie effectively lacking Sphagna) is chronicled in the macrofossil record of a number of sites (for example, Chambers *et al.* 2000), showing not only that peat formation can still occur beneath such vegetation, but also that it may simply represent a phase in the development of blanket mire. To-date much of the palaeoecological emphasis upon modified mires in Wales has focussed on those dominated by either *Eriophorum vaginatum* (M20) or *Molinia* (M25 species-poor variant), but work by Bostock (1980) on

the Berwyns demonstrated that forms of M19 lacking virtually any *Sphagnum* cover (and conforming to Tallis's (1969) Campylopus-Diplophyllum nodum) were a recent successor derived from the grazing and burning of vegetation with a much stronger *Sphagnum* element. This implies that the vegetation is at least potentially restorable."

JNCC (2009) [4-] in the Common Standards Monitoring guidance for upland habitats reviewed the effects of burning on upland habitats including blanket bog It was noted that fire"... is a major factor determining vegetation dynamics and the overall composition and structure of dwarf-shrub heaths. Burning can alter the vegetation composition, the pattern and age structure of plants, the carrying capacity for herbivores, and the associated fauna, and can alter the physical structure, nutrient status and even hydrology of soils and peat (for example, MacDonald 2003, MacDonald *et al.* 1998; Backshall *et al.* 2001, Tucker 2003). The impacts of fire are dependent on the intensity, frequency and scale of the burns, the type of vegetation burnt, and also the weather and soil conditions at the time of the burn. Too-frequent fires, intense fires and fires covering large areas can all be environmentally damaging. Fire can both increase and reduce biodiversity interest. Effects vary according to the intensity, seasonality, frequency, and size of fires (see SEERAD 2001b, now revised, Scottish Government 2011b)." "Views on the application of burning on blanket bog are diverse and indeed often highly polarised. Some people will cite stratigraphical studies of peat deposits as providing evidence that burning has occurred at intervals throughout the development of the peat profile; fire does not inevitably result in the destruction of bog communities nor does it typically bring an end to peat accumulation. Five points should be noted: a) bogs do not need to be burnt to retain their interest; b) there should be a general presumption against burning of blanket bog; c) it is possible to burn blanket bog which is dominated by *Calluna* without damaging the interest; d) there should be a presumption that degraded bogs can be restored until the contrary is demonstrated; and e) inappropriate burning can lead to erosion (including organic soil and carbon losses) and the loss of peatland habitats." "Human activities, primarily burning and draining, can result in changes to the hydrology and properties of the peat especially at the surface. Along with grazing, these activities can result in floristic changes to the blanket bog and valley bog communities. These floristic changes are usually regarded as degradation because they result in the loss of characteristic species of the bog communities and dominance of a few species of graminoids or dwarf-shrubs. The derived communities are poorer in species than the original communities." Blanket mire "communities can be heavily influenced by management, notably burning and grazing, leading to degradation and replacement by heath, mire and grassland NVC communities": H9 and H12 dry heath, M15 and M16 wet heath M25 *Molinia* mire and U6 *Juncus squarrosus* grassland.

Littlewood *et al.* (2010) [4-] reviewed peatland biodiversity in one of a series of reviews for the IUCN UK Peatland Programme's Commission of Inquiry. They noted that "prescribed burning can significantly alter vegetation assemblages and reduce the amplitude of surface patterning features (Hamilton *et al.* 1997). There is conflicting evidence on the form that such changes to the vegetation take. For example *Sphagnum austinii* was formerly a major part of some peatland systems at the mesotope or macrotope scale (Chambers *et al.* 2007) but its demise in the peat record coincided with an increase in burning activity. Its revival in recent decades at Cors Fochno, Wales, was linked to the control of burning activity on that reserve (Bailey 2003). However Shaw *et al.* (1997) found no firm evidence that managed burning results in long term damage to *Sphagnum* cover of blanket bogs. It has also been argued that prescribed burning leads to a dominance of heather and a reduction in other species (McVean & Ratcliffe 1962) although a systematic review of the impacts of prescribed burning on blanket bogs found that there was a tendency for burning to cause an increase in bryophytes and bare ground and a switch from ericoids to graminoids (Stewart *et al.* 2005, above)." "Burning on blanket bogs is now discouraged (Defra, 2007; Welsh Assembly Government, 2008; Scottish Government, 2011a) and there is little evidence to recommend otherwise. A longer cycle of burning is permitted on some drier heather-dominated peatland sites in England under the relevant code (Defra, 2007). However most such sites are likely to represent modified vegetation that is actually perpetuated by the continuation of burning and the burning cycles needs to be broken for peatland vegetation to recover."

Lunt *et al.* (2010) [4+] reviewed peatland restoration in one of a series of reviews for the IUCN UK Peatland Programme's Commission of Inquiry. They noted that "considerable gains to biodiversity

and reductions in carbon loss have been made over large areas by reduction in grazing animal stock densities and removal of burning management" but that "agreed burning continues on degraded peat bogs, where through regular burning, heather has gained dominance. Where carried out, this practice will continue to discourage *Sphagnum* and encourage the growth of tolerant vascular plants." "Vegetation changes in response to hydrological changes are also heavily dependent on burning and grazing management regimes. It is likely that on dry peatlands, dominated by *Calluna*, change to vegetation will not occur or only occur very slowly unless burning management is removed … In these heather monocultures it is also likely that the re-introduction of peat forming plant material will be required." They concluded that "managed burning and restoration of surface rich *Sphagnum* layers on deep peat are probably not compatible."

IUCN (2011) [4+] in a briefing note on burning and peatbogs to accompany the Commission of inquiry on peatlands report (Bain *et al.* 2011) reviewed the practice and effects of burning on peatlands. In relation to vegetation, they noted that "poorly managed 'hot' burns can have severe damaging consequences for peatland ecology, hydrology and soil processes. Such burning can degrade bog habitat, with reductions or loss of key bog species and structural diversity and can encourage more typically heath species (Ratcliffe 1964; Rowell 1990). Recovery back to bog species after a fire depends on the frequency and intensity of the burn along with other factors such as the condition of remaining bog vegetation, water levels, livestock numbers and altitude. It has been suggested that 'cool' burns, under the right controlled conditions, and may be compatible with the initial stages of peatland restoration management while rewetting takes place." In relation to this, "there are few studies on the benefits and practicalities of burning over other techniques such as cutting or layering (Lunt *et al.* 2011)."

Worrall *et al.* (2011a) [4+] reviewed burning on peatlands in one of a series of reviews for the IUCN UK Peatland Programme's Commission of Inquiry. They noted that "There are contrasting points of view with respect to the impact of fire on plant biodiversity and this to some extent depends on: (a) what is considered to be the historic vegetation; (b) the current vegetation, and indeed the vegetation that is to be conserved under existing conservation legislation; and (c) the potential target for the future." Regarding 'fire-sensitive' species, "there are published works, based on observational accounts, that some species are intolerant of fire and are damaged or even extirpated from a site after burning (McVean & Ratcliffe 1962, Rodwell 1991, Wiggington 1991, Stewart *et al.* 1994, Page 1997, Preston *et al.* 2002 and Ratcliffe 2002). However, many of these reports, implicating prescribed burning in the decline of species, are based on anecdotal evidence and observational correlations rather than experimental evidence and many of these references also acknowledge that other factors such as overgrazing, pollution and particularly drainage may be involved. Moreover, there is often a lack of detail on whether observations are in response to prescribed burning or wildfire." Regarding current vegetation, "some have argued that the current burning regime reinforces a *Calluna*-dominated vegetation that is relatively low in species (Lindsay, 2010; McVean & Ratcliffe, 1962; Rodwell, 1991). On this sort of modified vegetation, the current burning management is applied mainly for grouse and sheep, but at the same time it maintains a vegetation of low biomass relative to more mature phases of growth, and hence fire hazard. Where burning is not implemented the available fuel may increase and will likely have a high burn severity when wildfire strikes. Where this occurs the ecosystem response will also be severe as has been shown in the dry *Calluna* dominated heaths of the North York Moors after the 1976 fires (Maltby *et al.* 1990). However, this may not apply in blanket bog where the layering and growth of bog mosses may limit the available above-ground fuel. Additionally, not all wildfires are necessarily more damaging that prescribed burns and low severity wildfires can re-establish very quickly with little apparent damage to the ecosystem (Clay *et al.* 2010c).

It was suggested that future ecosystem recovery "will depend on the current and previous vegetation type and condition along with a range of other factors such as grazing regime and local pollution. Studies in the Peak District (Harris *et al.* 2011b) have shown that on *Calluna*-dominated blanket bog managed by prescribed burning there is a flush of species (vascular plants, mosses and lichens) in the immediate post-burn phase. Thereafter a long-term reduction in species diversity… after burning, and that if left unburned for 50 years there is no evidence of colonization by new species. Moreover, there are very few propagules of plants or bryophytes in the surface soil and litter. Thus, on these

admittedly degraded peatlands there is little hope of restoring more diverse communities without substantive intervention. However, this may not be typical of the rest of the UK situation. Such interventions might include wetting the peat by gully and grip blocking and the addition of new species. However, all of this would have to be carried out against a potential threat of summer wildfire. Other areas many not be as degraded as these Peak District sites, and where there is a more diverse flora of peat-forming species then maintenance of these mire communities or their restoration may be easier. *Sphagnum* species are key peat-forming species; however, there is little research into the effects of fire on their survival and recovery. *Sphagnum* may survive low intensity burns (Hamilton, 2000), however, in severe wildfires such as those on the North York Moors in 1976, *Sphagnum* species may be removed entirely (Maltby *et al.* 1990). In their study of burning and grazing at Moor House, Rawes and Hobbs (1979) showed that *Sphagnum* did recover following burning following a period of *Eriophorum vaginatum* dominance. Burch (2008) shows for wet heath sites of the North York Moors, the abundance of *Sphagnum* species was greatest at canopy heights of 30 cm, though these canopy heights reflect a variety of ages and stages of *Calluna* development. It is currently untested whether repeated cycles of burning leads to replacement of *Sphagnum* species, though hummock formers may survive fire better due to their higher moisture retention (Peatscapes Project, 2008). *S. tenellum* (Lindsay & Ross, 1994) and *S. compactum* (Okland, 1990; Slack, 1990) have been shown to be colonisers of burnt areas." The review concluded that "prescribed burning can bring positive and negative effects for a range of ecosystem services" including biodiversity. "For example, burning has been observed to increase grouse and sheep production; however, burning has also been noted to negatively affect the presence of some flora and fauna species, for example, meadow pipit (*Anthus pratensis*) and some *Sphagnum* species."

Fauna of upland peatlands

Reviews on invertebrates

Tucker (2003) [4+] reviewed the effects of burning on invertebrates of upland habitats in general and upland peatlands specifically. It was concluded that, "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 to come to any conclusions on this subject." Nevertheless, referring to the work of Dennis & Eales (1997, 1999) and Eales & Dennis (1998) he notes that the presence of the declining large heath butterfly is weakly positively correlated with burning, but weakly negatively correlated with severe burning. It was suggested that this may be due to the larval foodplant, *Eriophorum vaginatum*, tending to be favoured by "moderately frequent" burning (Hobbs 1984) and that it may counter succession on drier sites and favour *Erica tetralix*.

Grant *et al.* 2012 [4+], as part of a wider review on the effects of grouse moor management, reviewed the impacts of burning on invertebrates. It was concluded that although "rotational muirburn on dry dwarf shrub heath increases the diversity and abundance of some invertebrate groups (notably ground beetles and spiders) and appears to have the potential to increase the overall diversity of terrestrial macro-invertebrates at least, though the resultant increase in structural diversity of the vegetation." However, "it is unclear whether such potential benefits arise on wet heath and blanket bog, although a carefully managed, 'sensitive', rotational muirburn regime on such habitats is expected to increase the abundance of at least one species of conservation importance (the large heath butterfly). Some limited evidence suggests that muirburn may reduce the abundance of soil invertebrates but does not affect the overall abundance or taxonomic richness of aquatic invertebrates in moorland streams, although it may reduce the abundance of some mayfly and stonefly species."

Review on reptiles

Glaves *et al.* (2005) [4+] in a wider review of the effects of burning on biodiversity, soils and hydrology, reviewed the effects on herpetiles though this covered moorland and heathland habitats in general. Nevertheless, two UK BAP species, adder and common lizard, are associated with upland peatlands. It was noted that "few species of amphibians and reptiles (and no obligate upland species)

occur in the uplands (Coulson *et al.* 1992), and Tucker (2003) was unable to identify any scientific studies on the impacts of burning on them. More work has been carried out on the ecology and impact of management on herpetiles in lowland habitats, especially heathland. Most reptiles require open, sunny areas, hence land management, including burning, can have an important role to play in halting succession and maintaining suitable conditions. However it can be damaging. Responses to a Froglife/English Nature questionnaire survey from 471 sites indicated that fire was considered a negative factor for adders and slow-worms *Anguis fragilis* on some sites (13 sites, 23% of sites where factors were identified, for adder and 11, 28%, for slow-worm) (Baker *et al.* 2004). Generally the increase in temperature as a fire passes over only extends a few cm below the surface. Thus, depending on intensity and timing, reptiles may survive the direct effect of fire below ground in winter hibernacula and even summer refugia." "The common lizard Lacerta vivipara is probably the most abundant reptile on moorland and heathland. It is unlikely to escape when moorland or heathland is burnt, at least when it is not hibernating (February/March–September/October, Beebee & Griffiths 2000). Frazer (1983) suggests that, when below ground, lizards may survive rapid cool burns, though they remain liable to predation on bare post-burn areas. Simms (1969) studied recolonisation of a burnt area of (lowland) heathland at Strensall Common, Yorkshire. Though hibernating at the time, the fire killed all the lizards (presumably due to a hot burn), though they were still present on the periphery. They did not start to use the burnt area until the second year after the fire and the population did not rise to pre-burn levels until another year later. Since it is widespread, and occurs frequently on areas that have regularly been burnt, controlled burning seems unlikely to have a critical effect on the distribution of this species, though extensive fires might cause losses from smaller sites (which are more common in the lowlands)."

"The adder is locally common on moorland apparently including in some areas that are regularly burnt, but is also absent from much moorland and heathland, including much of the Pennines (Frazer 1983). It has been suggested that burning (and overgrazing) may have eradicated adders from some moorlands (for example, Offer *et al.* 2003) and Whiteley (1997) suggests that it may be a factor in the absence of adders from much of the Peak District moorland. However, the Panel are not aware of any evidence to confirm this. It is more widespread in various habitats in the lowlands. Adders hibernate for about five months between mid October and late February/March, slightly longer for females, with hibernacula usually located in areas of thickly vegetated dry ground (Beebee & Griffiths, 2000), sometimes well below ground (up to 2 m; Viitanen 1967). According to Wild & Entwistle (1997) exact timing and duration of hibernation varies regionally with latitude, altitude and annual variation in climate. However, Whiteley (2003) lists records of early emergence on Peak District moorland between mid-February and early-March (similar to southern England) when it had previously been considered to occur later from mid-March. When active, adders are likely to be killed by fires, though they may survive below round in hibernacula (and possibly even in summer refugia) depending on the intensity of the fire. Thus, Wild & Entwistle (1997) recommend that 'moorland should not be burnt before November or later than mid-February because Adders, especially on sunny days, may be burnt'. Nevertheless, Simms' (1972) studies in the same burnt area of heathland where he studied (common) lizards (above) showed recolonisation within a few years. No adders were present until the vegetation had re-established and lizards had recolonised; they then gradually spread back across the area over three–six years."

Reviews on breeding birds

Brown & Bainbridge (1995) [4-] reviewed the ornithological importance and value of grouse moor management for upland birds. They did not specifically address burning effects or peatland habitats separately from other moorland, but noted that "grouse moors have been instrumental in protecting uplands from forestry and agricultural intensification, but important populations of many upland species are found on moorland managed for other purposes. Very few, if any, species are dependent on grouse moors per se." They also noted a then paucity of quantitative studies of habitat selection or of the effects of moorland management on species other than red grouse.

Pearce-Higgins *et al.* (2009) [4+] reviewed the international importance and drivers of change of upland bird populations particularly in relation to changes in moorland management. On moorland in general rather than specifically peatlands, they suggested that 2 species requiring some short, open

vegetation will probably benefit from rotational muirburn; and indeed, both curlew and golden plover nesting on grouse moors select recently burnt areas for nesting, and for the broods to forage (Robson, 1998; Whittingham *et al.* 2000, 2001). Furthermore, the association of grouse moors with heather vegetation (Robertson, 2001) means that species associated with some heather in the landscape, such as black grouse, merlin, hen harrier, short-eared owl and ring ouzel, tend to be more abundant on grouse moors (Thompson *et al.* 1997)." On the other hand "there is little evidence for benefits of grouse moor management to moorland passerines, with meadow pipit densities reduced on grouse moors, possibly linked to detrimental effects of muirburn (Smith *et al.* 2001; Tharme *et al.* 2001). Putative relationships from other studies of passerine density and grouse moors probably result from associations with either heather (stonechat and ring ouzel) or grass cover (skylark and wheatear)." They suggest that "if management of blanket bog to improve carbon storage, water quality and retention (for example, re-wetting by drain blocking) increases habitat quality and invertebrate abundance, it may benefit upland birds. However, "if there are reductions in grazing and rotational heather burning, this is likely to be detrimental to those species dependant on short, open swards, although other bird species are likely to benefit from such change in the longer term as scrub and woodland may develop through vegetation succession. Therefore land-use changes that are likely to occur will represent opportunities for bird populations in the uplands as well as threats."

Sotherton *et al.* (2009) [4+] reviewed the effects of managing the uplands for game and sporting interests from an industry perspective. This included the large-scale study by Tharme *et al.* (2001, see above), local studies in the Peak District by Daplyn & Ewald (2006, see above) and Pearce-Higgins *et al.* (2006, see above) and then unpublished data from the GWCT's Uplands Predation Experiment (Fletcher *et al.* 2010, GWCT 2010), see above).

Referring to differences in findings in studies on the impact of grouse moor management on changes in moorland breeding birds in the Peak District between surveys in 1990 and 2004, the authors suggest that these may reflect differences in the measures of grouse moor management: Pearce-Higgins *et al.* (2006, see above) used density of burning, whilst Daplyn & Ewald (2006, see above) used two measures, burning and presence of a gamekeeper; and also the greater detail of classification of habitat in the former. The latter found more positive relationships with grouse moor management than the former, though it was noted that "both of the analyses suffer from the fact that the surveys are snapshots in time, making comparisons between them difficult" (see also Grant *et al.* (2012) below).

The authors note that "neither the Peak District surveys nor our experiment address how well waders are doing at a much the wider scale at which grouse management is undertaken" but they then review changes in distribution between successive national breeding bird atlases (Sharrock 1976, Gibbons *et al.* 1993) and show large area differences. Three ground nesting birds, red grouse, dunlin and golden plover, showed much lower percentage range declines in the north of England where grouse moor management is concentrated than in SW England and Wales where it is not (though it should be noted that the other two areas have much lower numbers of the species concerned at the edge of their range so other factors may be involved). The authors note that "the presence of grouse moors is not the only determinant of wader numbers in the UK, nor is the lack of grouse management the only factor causing the decline of waders throughout the UK (Pearce-Higgins *et al.* 2009, see above). There is evidence for wader decline in the North Pennines where grouse moors predominate and for wader increase in areas such as the Outer Hebrides where grouse moor management is absent (Sim *et al.* 2005, see above). However, the impact of grouse moor management can only be addressed when accurate maps of grouse moors are available or detailed management of wader survey squares has been quantified. Such data rarely exist, are so such detailed analyses are rare."

Grant *et al.* (2012) [4+] carried out a detailed review of the costs and benefits of grouse moor management to biodiversity and aspects of the wider environment including the effects on moorland birds. It was noted that burning may affect moorland birds in two main ways: "by direct destruction of nests and through modification of habitat (including the structural characteristics and the availability and condition of both plant and invertebrate food sources." On the former, it was noted that "the available data suggest limited overlap only between the date on which generally has to cease and the time of clutch initiation of many ground nesting birds on moorland (Glaves *et al.* 2005, Moss *et al.*

2005, both see above). Amongst the ground nesting species that show greatest overlap in this respect, several select nest-sites in short vegetation (notably golden plover, lapwing and curlew – Whittingham *et al.* 2002, Robson 1998), where burning is unlikely to occur in grouse moors, but others select nest-sites in tall *Calluna*, where muirburn is likely. Short-eared owls and stonechats may be amongst the most vulnerable species in this respect, given that a substantial proportion of nesting attempts are initiated before 15 April ... and that both species nest in tall *Calluna* (Roberts & Bowman 1986, Moss *et al.* 2005, see above, Pearce-Higgins & Grant 2006, see above)."

Appendix 6 The studies and evidence for the effects of burning on carbon sequestration

Hard Hill plots and related Moor House NNR carbon budget studies

Garnett *et al.* (2000) [1+] used the long-term Hard Hill plots at Moor House NNR in the North Pennines, set up in 1954, to determine whether sheep grazing and burning of moorland every ten years influenced carbon accumulation in peat. A chronological marker using spheroidal carbonaceous particles (from industrial pollution assumed to affect the whole area equally) was used to establish a baseline for comparison of accumulation and was validated by use of charcoal profiles assumed to be produced by local burning.

No differences in the accumulation of peat between grazed and ungrazed treatments were detected. This may be due to the very low sheep stocking rates at the site (reported as 0.02-0.2 sheep ha^{-1} in summer only though other studies give slightly different figures). No effects on above ground biomass have been detected through these rates in other studies at this site. Significant differences in carbon accumulation between the 10-year burn and no burn plots were found (3.1 +/- 0.4 Kg m^{-2} vs 5.4 +/- 0.6 Kg m^{-2}). It was not possible to say whether this resulted from reduction in the rate of accumulation and/or reduction in carbon stores through burning accumulated peat. The differences between treatments reflect the impact of burning over a 32-year period and the difference in carbon sequestration in that period represents 73 g m^{-2} yr^{-1}.

Garnett *et al.* (2001) [2++] estimated the stored carbon among an extensive range of vegetation and soil types in 22 1 km^2 land units contained entirely within the boundary of the Moor House NNR using: existing digitised data of vegetation extent; equivalent true surface area allowing for slope; above ground biomass data obtained from previous detailed studies and converted to carbon (C) content using the accepted 47% C-value; field sampling of soil C content with particular emphasis on blanket bog. Missing data for minor vegetation and soil types were obtained from similar locations in the UK or values derived from similar classes at Moor House. An empirically-derived correction factor was applied to peatland soils which were prone to compaction during sampling. True and 2D surface areas were compared to determine the potential magnitude of error when stored-C is based on the latter alone. Stored C was calculated for the full range of vegetation and soil types as well the 22 x 1km^2 land units, which could then be compared to those of the national inventory.

Results indicated that soils are more important than vegetation in terms of C storage and reiterate the importance of peat soils in particular. Whereas figures for the total amount of C in vegetation were similar to those of the national inventory, the figures for soils derived from the study were more than three times lower. Differences were attributed to the much higher resolution of the soil C content data at Moor House NNR and the more representative calculation of C content for soil complexes; current national and global estimates of terrestrial stored C should therefore be treated with caution. Although the study does not relate directly to burning, it was included as it is relevant to other related studies at Moor House.

Ward *et al.* (2007) [1++] used the long-term Hard Hill experimental burn plots at Moor House NNR, set up in 1954, to study carbon dynamics in an upland peatland ecosystem subject to management by burning and grazing. Sampling of vegetation composition; carbon stocks in organic (O') horizon, F and H horizons and litter; peat microbial properties and N availability; trace gas flux (CO_2 and CH_4); and soil solution DOC were measured in a 2 x 2 factorial design (burned (10-year) and grazed, burned ungrazed, no burn grazed, ungrazed and no burn) over an 18-month period towards the end

of the 10-year burn cycle. Results show that long-term land management has significantly affected both surface storage and flux of C in peatland and that these effects are strongly related to changes in vegetation arising from burning and grazing. Management treatments have different effects apparently relating to the intensity of disturbance.

Burning and grazing reduced above ground C by 56% and 22% respectively and burning reduced C storage in the F and H horizons by 60%. This reduction was not detected in total ecosystem storage because the bulk of C is at depth. Overall regular burning (10-year rotation) led to a loss of 167 g C m^{-2} from the peat surface and 88 g C m^{-2} from above ground vegetation. The total loss of C due to burning is estimated to be 25 g C m^{-2} yr. Fluxes of CO_2 were affected by burning and grazing. Both gross CO_2fluxes of respiration and photosynthesis increased due to burning suggesting long-term acceleration in process rates. Burning also created a greater net sink for CO_2 by increasing photosynthesis at a greater rate than respiration. Grazing had similar but lower magnitude effects on CO_2 flux. Disturbance from grazing and burning therefore increases ecosystem process rate and gross CO_2 fluxes and reduces net CO_2 efflux. Though CO_2 fluxes were responsive to management the main factor controlling fluxes was seasonal change in climate. There was an interaction between seasonality and land use. This raises the potential for greater rates of flux due to global warming. Peatland fluxes of CH4 and DOC are much smaller than CO_2 but are still important. In this study fluxes of CH_4 were always positive (as expected in a wetland) and burning caused a small reduction in flux but grazing increased it. Burning did not influence DOC release in soil water in this study but grazing produced a small increase. Both CH_4 and DOC fluxes showed a seasonal pattern.

Clay *et al.* (2010b) [2++] present a carbon budget for the Trout Beck catchment at Moor House NNR based on data from the Hard Hill plots and the Environmental Change Network (ECN) sampling site in the catchment. Fluvial and gaseous fluxes are considered. Measures of DOC and depth to water table were taken from dipwells in unburned, 10 year and 20 year burn treatments including both grazed and ungrazed plots. Secondary parameters of carbon were estimated from these, environmental data and from established relationships reported in the literature. Data were collected in the period 2005-08.

The results show that all treatments were carbon sources with the unburned treatments being the largest source and burned treatments smallest. Unburned sites were on average a source of 156.7 g C compared to sources of 109.6 and 125.9 g C m^{-2} yr^{-1}on the 10 and 20 year burn plots. Differences are likely to be due to lower primary productivity in old vegetation and higher ecosystem respiration perhaps due to lower water tables in unburned sites. These results conflict with other findings (Worrall, 2007) that show that Moor House NNR is a carbon sink. Long-term fluxes of carbon are modelled including losses through combustion in management fires. There is a critical threshold at which losses of carbon in combustion are balanced by reduced losses through increased primary production and lower respiration. Assuming a biomass loss of 85% in fire, after 100 years the cumulative losses in the 10 and 20 year burn rotations were greater than if no burning had occurred. By extending the rotation to 25 years the effect of burning would be neutral. However, the authors suggest that at longer rotations fires are likely to be more intense and release more carbon. If total combustion occurs and burn cycles exceed 32 years then it is predicted that less carbon will be lost than if there is no burning.

Orwin & Ostle (2012) [1+] used the long term burning experiment on high altitude *Calluna-Eriophorum* blanket peat at Hard Hill, within Moor House NNR, to study the role of mosses in carbon cycling after fire. The study focussed on the moss species *Sphagnum capillifolium*, *Hypnum jutlandicum* and *Plagiothecium undulatum*, which have a wide ecological tolerance and which are ubiquitous among the burn treatment plots. Two patches of each moss species and two bare patches of peat were selected in each of the three burn treatments (no burn since 1954 control, 20-year and 10-year burn rotation) across the four replicate blocks at the site. Carbon dioxide flux was measured in one patch both in full natural illumination and with light excluded artificially. This provided an estimate of moss photosynthetic rate and gross respiration from which Net Ecosystem Exchange was derived. Total ‘photosynthetically active radiation’ (PAR) at the surface of the patch and soil temperature at 5 cm depth were also quantified. A mesh bag containing a standardised litter sample of *Eriophorum vaginatum* was buried beneath the surface in the second patch, where it was left for a

year. The loss in mass was determined, adjusting for initial moisture content. Litter from all three moss species was collected and allowed to decompose for 42 weeks within sealed petri dishes on top of peat samples but separated by fine mesh. Peat samples were obtained from the respective plots of origin for each litter sample and maintained within the petri dish at constant moisture levels equivalent to field conditions. Insufficient biomass was available from some plots for a fully balanced design. The C:N ratio of the litter was determined at the start and finish of the experimental period and the loss in mass was determined as for the in situ study.

At the time of measurement, the 10-year plots (1-2 post-burn) had low vegetative biomass and were dominated primarily by mosses and some graminoids, whereas biomass on the 20-year (12-13 year post-burn) and control plots was considerably higher and close to 100% cover. The moss species were primarily responsible for differences in the parameters quantified, with very few burn effects detected in either the in situ or ex situ study. In general, mosses were shown to be sources of CO_2 during the short period of the investigation but *Sphagnum* had the greatest propensity to sequester carbon in certain months; this could be attributed to periods when photosynthetic rate was higher and/or respiration rate lower than for other the species. Soil temperature at 5 cm depth was higher beneath bare peat and for 10-year burn plots, and PAR was highest for this treatment in June. The rate of litter decomposition was not affected by burn treatment or moss species. Litter characteristics differed among moss species with Plagiothecium having the lowest C:N ratio and decomposing most rapidly under controlled conditions. Litter of both *Sphagnum* and Plagiothecium lost mass and carbon faster on peat from no burn compared to burnt plots, whilst least C was lost by *Sphagnum* in the 10-year burn. Comparing initial and final C:N suggests that litter characteristics other than C:N ratio, and which affect the rate of litter decomposition, change in response to disturbance per se. Predicting the effect of fire on peatland C-cycling requires knowledge of the species composition of the community and the extent to which species traits affecting c-sequestration vary autogenically and allogenically.

Other carbon budget studies

Grand-Clement (2008) [2-] studied the effects of experimental/managed heather burning on blanket bog on soil organic matter and peat accumulation at Moor House NNR in the North Pennines and Howden Moor in the Peak District. The method involved collecting a series of surface soil samples, depth samples and peat cores. These samples were analysed for pH, moisture content, loss on ignition, total carbon and nitrogen content. An incubation experiment was carried out to calculate biomass C. ^{13}C NMR (nuclear magnetic resonance) spectroscopy was used to evaluate the different functional groups of the organic matter. Black carbon was measured using benzene polycarboxylic acids and chemical oxidation. Peat soil accumulation was measured by radiostopes.

The method used in the study (^{13}C NMR spectroscopy) showed no increase of aromatic structures in the bulk soil, either immediately after a burn or with longer fire rotations (ie every 10 or 20 years). The author suggested that the transformations taking place seemed to be related to the increased degree of decomposition of the soil organic matter. Charcoal production was low (<1 % of the biomass burnt) and on a spatial scale the carbon added to the soil was negligible (about 16 g C ha). The results indicated that where processes of organic matter modification occur, their impact remained negligible. Few changes were observed in black carbon (BC) quantities as a direct result of fire treatment. However, the soil still contained significant stocks of BC (for example, between 1 and 1.4 T ha in the top 5 cm in the Peak District sites). Relatively high peat accumulation rates were measured (between 4.2 and 14.1 cm/yr) with a high variability between plots. The results suggested it was not possible to conclude on the effect of fire on peat accumulation rates.

Farage *et al.* (2009, 2010) [2-] reported on an investigation of the carbon cycle in a burned grouse moor in the Yorkshire Dales carried out in 2003 and 2004. The sites sampled appear to been *Calluna*-dominated, degraded blanket bog, perhaps with transitions to heath; variously described as on soils with an organic layer rarely less than 30 cm and as peaty gley with a peat/organic layer 30-50 cm but with a range 10–80 cm. Biomass in heather stands was investigated at three ages post-burn: 1, 12-15 and c.25 yr. Below ground biomass, soil carbon (0.3 m) and soil respiration were measured and biomass remaining after burns and trace gas emissions (methane and Nitrous oxide) were estimated.

Carbon stored in heather biomass ranged from 600–1,325 g C m^{-2}. Mean above ground biomass in stands were 1,262g m^{-2} (<1 yr), 1,575 g m^{-2} (12-15 yr) and 2,757g m^{-2} (c.25 yr). (The first value was challenged by Legg *et al.* (2010) as being anomalously high and suggesting very low rates of combustion, as was the 5,000 g m^{-2} value below). Below ground storage was 2,000 g m^{-2}, 5,000 g m^{-2} and 2,000 g m^{-2} in the three heather classes respectively. Soil carbon stores averaged 9,951 g C m^{-1} across all sites (measured to 30cm). Losses due to burning are reported as 16 and 24% of above ground biomass in the two years investigated. This corresponds to 103 and 201 g C m^{-2} removed from the system which is considered to be neutral. The trace gas emissions for these events are estimated to be equivalent to 20 and 39 g C m^{-2}. Soil respiration was measured at a mean of 0.281 g CO_2 m^{-2} hr measured in April 2004 (soil temperature 7.6o C). Legg *et al.* (2010) presented reservations about this work relating to sampling procedures and analyses. Some of the data were considered anomalous and difficult to explain, though they suggested that sources of variation between sample sites may not been eliminated or accounted for. Sample sizes were small and sampling protocols and rationale not fully explained.

Clay & Worrall (2011) [2+] investigated the biomass and carbon losses following a moorland wildfire in the Peak District. The vegetation of the local area was dominated by heather, bilberry, and cottongrasses, with areas of *Sphagnum,* on deep peat. Changes in above ground carbon stocks were calculated using a combination of field data, laboratory measurements and literature values. Vegetation monitoring showed that dwarf shrubs and grasses occupied c. 46% and 33% respectively of survey quadrats in unburnt sections, while similar areas were occupied by char and exposed soil in burnt sections. Mosses (including *Sphagnum)* occupied a similar percentage of quadrat area (c.5%) in both burnt and unburnt areas, suggesting that mosses were little affected by the fire. The authors suggested that the high water content of mosses resulted in them merely drying out rather than burning in low temp fires.

The results showed that approximately 14% of the original carbon in the original above-ground biomass remained on the site after the burn. The authors suggested that loss of carbon resulted from conversion to gaseous products during combustion (CO, CO_2), formation of airborne particles, and erosion of char from the surface following the fire. Black carbon (char) production was approximately 6 gC m^{-2} which constituted 4.3% of the biomass lost. Although black carbon may degrade over time, it has a very long mean residence time. The paper concluded that char production during moorland wildfires could be an important carbon sequestration mechanism to peat soils in the long-term but on short-medium term scales, fires lead to net carbon emissions. The authors noted, however, that if the fire ignites the peat, then any long-term carbon benefit from the production of char will be negated as the majority of the carbon stock in UK peatland is held within the peat rather than the vegetation. Factors considered likely to influence char production were variation in fuel types, microtopography of the peat surface, local weather conditions during wildfires, and fuel moisture content.

Harris *et al.* (2011a) [2+] considered fire characteristics and vegetation biomass reduction in 17 (‘pressurised fuel assisted’) ‘cool’ management burns (in March and April) on Howden Moor in the Peak District (described further under sub-question e).

The results showed that pre-burn mean dry biomass was 301g. Maximum temperatures recorded were 982°C and 993°C at base and canopy, respectively. Burn residence time ranged from 11-547 seconds at the base and 6-474 seconds at the canopy. Combustion was incomplete in all burns. Up to 93% of total dry biomass was removed (mean 66%, range 21-93%) and up to 96% and 97% of litter and *Calluna* dry biomass, respectively. Thus, there was considerable variation in fire characteristics and biomass measurements, by 1-2 orders of magnitude, which was greater within than between burns.

Allen *et al.* (2013) [2+] modelled the impact of varying managed burning rotation intervals on above-ground fuel load and released carbon in comparison to wildfires of varying return intervals using data from a species-poor *Calluna*-dominated (M19/M20) blanket bog at Howden Moor in the Peak District. Fuel load accumulation data were obtained from a chronosequence study (Harris *et al.* 2011b [2+], above) and combustion completeness data from prescribed burning experiments (Harris *et al.* 2011a [2+], above). The stable age structure of vegetation under varying burn rotations was predicted using

an age-structured matrix model and moorland above-ground fuel load calculated using a boot-strapping approach. Long-term carbon losses were predicted under varying wildfire return intervals. The model did not consider below-ground carbon or the effect of managed burning on wildfire probability.

There was a clear interaction between managed burning rotation interval and wildfire return interval. At 50- and 100-year wildfire return intervals, carbon losses were minimised by short prescribed burning rotations. However, under a 200-year wildfire return interval, carbon loss was minimised by long rotation intervals where delayed regeneration was modelled. Under a 50-year wildfire return interval, 8-year prescribed burning rotation intervals could reduce carbon loss by 22% or 34% compared to 25- and 50-year rotations, respectively. It was concluded that the modelling approach "provides a first approximation to the above-ground carbon balance between prescribed burning and wildfire frequency at a single site. At our study site, long prescribed burning rotations may minimise carbon loss at low wildfire return intervals. However, if wildfire incidence increases, more frequent prescribed burning is likely to minimise overall carbon loss. Well-informed prescribed burning on a short rotation may produce smaller carbon losses than longer rotations under future climate conditions." However, in peatland habitats such as the study site, the amount of carbon in the peat greatly exceeds that in above ground vegetation (Gray & Levy 2009).

Peat accumulation

Only one relevant study was identified on peat accumulation in addition to those described above at Moor House NNR (one of which included work at a second site in the Peak District).

Pietikäinen *et al.* (1999) [2+] determined fire history and the effect of fires on the long-term (apparent) rate of carbon accumulation (LORCA) in the Patvinsuo National Park raised mire complex in eastern Finland using stratigraphic and pollen analyses on 98 peat cores.

The peat cores were characterised by a large number of charcoal layers and the age of the basal peat varied between 57 and 10,500 years. Mire fires slowed down the progress of vertical peat accumulation and resulted in "great carbon losses". The average LORCA in the Patvinsuo cores (9.2 ± 1.0 (SE) g m^{-2} yr^{-1}) was lower compared to the average for all mires in the southern half of Finland (17.7 ± 0.6 (SE) g m^{-2} yr^{-1}). The average rate of carbon loss in the Patvinsuo mires was 9.5 ± 1.0 (SE) g m^{-2} yr^{-1} and the mean carbon loss in an individual fire was estimated to be 2.5 kg m^{-3}.

Erosion

Kinako & Gimingham (1980) [2+] studied the magnitude of soil loss through management burning, the effect of slope angle on this, and the effect of vegetation recovery in reducing habitat deterioration. Two study sites were selected in NE Scotland where soil types were described as peaty podsols, and vegetation communities were dominated by *Calluna-Vaccinium*, with associated species including *Erica tetralix* perhaps indicating wet heath. Soil erosion was measured using iron 'erosion stakes' and vegetation recovery was measured as rooted plant frequency and height. Analysis was carried out for the effect of vegetation recovery using regression equations and correlation coefficients.

Results showed that reductions in level of soil surface occurred at all sites (0.27-0.55 cm). No consistent relationship with slope angle was identified. Initial rate of erosion positively correlated to slope ($r = 0.93$) at only one of the two study sites. A uniform pattern in relation to time was shown, with erosion reaching a maximum within eight months of the burn, and complete stability restored after 15-20 months. Habitat deterioration though soil erosion was least where vegetation recovery was most rapid, although levels of significance varied ($p = 0.001$–$p = 0.05$). The authors suggest the impact of soil loss may be greatest where large areas are burnt, precluding adequate restoration of material from surrounding areas.

Shelter *et al.* (2008) [2+] studied the relationship between *Sphagnum* cover, organic soil depth and soil organic matter stocks. The study was carried out in Alaskan black spruce forests, with a

Sphagnum ground flora. Ground layer moss/lichen composition, soil organic layer depths and organic matter concentrations were recorded along a series of transects in burnt and unburnt forest (presumably in wildfires) at two study sites. A total sample size of 25 points per site was used.

Results of the study showed that organic soil depth varied with burn status (ie burned/unburned) ($p < 0.0001$) and surface fuel type ($p < 0.0001$). Soil organic matter stocks showed a significant difference between burn status ($p < 0.0001$) and surface fuel type ($p < 0.0001$). Burning was shown to reduce organic soil depth by 55% and organic matter stocks by 36%. Microsites dominated by Sphagna had more than a twofold greater soil organic matter stock than microsites dominated by other ground vegetation.

Peat pipes

Holden (2005a, 2005b) [2+] studied the relationship between pipe frequency and vegetation cover type in upland blanket bog peat catchments, and demonstrated that *Calluna* species are one causative factor of piping in blanket peat catchments. The frequency of soil pipes in the 160 plots was evaluated using ground penetrating radar (GPR) and the influence of land management (moorland gripping), topographic position, slope gradient and aspect on the frequency investigated using GLM.

GPR on 960 plots over 160 British catchments demonstrated that piping was prevalent throughout blanket peats with a mean of 69.2 per km of GPR transect and was more frequent where bare peat (mean = 149 pipes/km) or *Calluna* (87 pipes per km) were present compared to other species (67 pipes per km). An additional case study of 32 plots at Moor House NNR in the North Pennines, where it was possible to control for topographic index, peat depth, and water table, provided additional supporting data, piping was greater under the *Calluna*-covered peat than under other vegetation covers. Further, laboratory experiments demonstrated that the proportion of water flowing through macropores almost doubled following the ten years of rainfall delivered during the 140 day experiment to recently colonised *Calluna* blocks. The author suggested that the development of *Calluna* woody stems and root structure allows preferential flow development and that this could increase the initiation of soil pipes. In contrast, bare peat blocks did not suffer increased piping under experimental high rainfall conditions. This may be because they had their top layers removed prior to the study starting and therefore there was no initial desiccation. Holden suggests that surface desiccation during dry summer periods could be a pre-requisite for pipe development in bare peat. Thus the study demonstrates that bare peat and woody *Calluna* plants are associated with increased levels of subsurface piping, potentially increasing subsurface erosion, carbon loss and chances of gully development.

Wind speed and air temperature

Fullen (1983) [3+] assessed the effects of heather removal by burning on moorland air temperatures and wind velocities at a *Calluna*-dominated site on peat in the North York Moors. Temperature and wind velocity were recorded above a mature (37 cm tall) *Calluna* stand and above the surface of a recently burnt (in April 1978) *Calluna* stand. Temperature was recorded using an automatic recorder between September 1978 and August 1979 (348 days) and wind velocity measurements were taken over an eight-hour period in June 1980.

Temperatures above the burnt moor were extreme and variable at instantaneous, diurnal and seasonal timescales. Differences in daily maximum temperature, daily minimum temperature, daily maximum temperature variation, daily duration of sub-zero temperatures and daily number of freeze-thaw cycles were all significant ($p < 0.001$) as were differences in daily mean temperature ($p < 0.05$). Temperatures were higher on the burnt stand during summer, but higher in the *Calluna* stand in winter).. Burnt moorland was reported as being more susceptible to freezing temperatures and more prone to freeze thaw cycles (100 during the year c.f. 64 in the heather stand). *Calluna* stands act as a protective buffer against temperature extremes, removal of which exposes moorland to temperatures which may promote breaking and fragmentation of peat surfaces.

Wind velocities at three heights above the burnt moorland were higher than above the heather stand. Wind velocity was significantly different from the *Calluna* stand ($p < 0.001$) at both 9 cm and 41 cm above surface height ($p > 0.001$), but not at 100 cm. The author suggested that these results will lead to enhanced development of desiccation crack, fragmentation of peat surfaces and therefore enhanced wind and water erosion.

Methanotroph activity

Chen *et al.* (2008) [2++] studied the impact of burning on the diversity and activity of methanotroph bacteria (micro-organisms that utilize methane) in peatland soils. The study was undertaken using deep peat soils from the Hard Hill plots at Moor House NNR using four soil cores from each block. In addition, ten monoliths were cut from unburned peat and five left with vegetation and five had all vegetation including roots removed. They were tested for pH and water content on all soils and the potential CH_4 oxidation capacity of the different soils at different depths. Methanotroph community structure and total bacterial diversity were measured.

The pH and water content of the soils whether burnt, unburnt, vegetated or bare were found to be broadly similar. The most active region for CH_4 oxidation capacity for most soils was 5-10 cm depth, although two of the four four unburnt samples were most active at 10-15 cm depth. The measured CH_4 oxidation potential was c.25µmol g-1 day-1 with no difference between burnt and unburnt treatments. 'Type II' methanotrophs were the most abundant in all soil samples. However, the relative abundance of certain 'type I' methanotrophs was higher in unburned treatments. Total bacterial diversity was similar in all soils, though there was some separation of burned and unburned treatments. This was also compared with nearby grass-dominated soils and this suggested a correlation between bacterial community profile and plant cover. Removal of *Calluna* vegetation decreased the CH_4 uptake potential. Potential CH_4 oxidation activity of soil with *Calluna* was significantly ($\alpha=0.01$) higher than when *Calluna* had been removed. Vegetated soil had about five times higher bacteria/methanotrophs than non-vegetated soil. Both vegetated and non-vegetated soils had similar community make up of methantrophs, again dominated by type II methanotrophs. The results indicate that the methanotroph community is influenced by above-ground vegetation cover. Burning did not appear to affect the CH_4 oxidation potential, but may cause subtle differences in the ratios of different elements of the methantroph community. Removal of *Calluna* cover did not appear to influence the community make up but did cause a uniform decrease in the population size.

Modelling

Worrall *et al.* (2010a) [2+] proposed a method for assessing the probability that land management interventions will lead to an improvement in the carbon sink represented by peat soils. The study reviewed the literature for information on annual carbon budgets using a meta-analysis. The number of studies with a positive outcome was expressed as a percentage of all studies with a definitive outcome. The effect of individual components of the carbon budget were added together to calculate an effect on the total carbon budget using a weighting rule derived from the stoichoimetry of the carbon budget of the Moor House study site in previous research. Using this method the study considered a range of management interventions, including managed burning.

The study showed that managed burning would bring a carbon disbenefit compared with cessation of burning. A 7% probability of a managed burning carbon budget benefit was calculated (and a 40% probability of a GHG benefit). There were no studies included of cessation of burning, but the study assumes the opposite of managed burning, ie a 93% chance of carbon budget benefit from cessation of managed burning.

Couwenberg *et al.* (2011) [2-] assessed GHG emissions from peatlands using vegetation as a proxy. The study did not directly consider the impact of burning, but it is indirectly related through occurrence of different vegetation types. The study was based on the current scenario (2009) of vegetation dominated by *Calluna*, *Eriophorum vaginatum*, Polytrichum strictum, *Sphagnum* species and *Drosera rotundifolia* in wetter areas and presence of *Betula* spp. on a raised bog in Belarus. Scenarios for 2039 were: a baseline scenario (with *Betula* spp. expansion favoured by the presence

of *E. vaginatum* tussocks leading to areas of 'forested bog-heath'); and a project scenario (wet *Sphagnum* communities expanded at expense of bog heath with impaired growth of dwarf shrub and trees).

Current GHG fluxes were estimated to be 5,471 t/yr CO_2-eq. (average 7.8 t/ha/yr). Results for the baseline scenario estimated GHG fluxes of 5,527 t/year (7.9 t/ha/yr), and for the project scenario, 2,403 t/year (3.4t/ha/yr) (an estimated emission reduction of 3,124 t/yr (4.5 t/ha/yr) in 2039). While these estimates were based on the best available data, and assumptions made for the purposes of the study, reliable data for GHG flux is limited and available publications are poor in the description of site conditions.

Critical synthesis on carbon balance

Lindsay (2010) [2+] in a critical synthesis on carbon and peatbogs provided "a critical examination of various key papers which have increasingly come to inform the debate about peat and carbon in recent years." This included a discussion topic on burning and peat bog systems. One of the general issues identified was the sometimes insufficient or misleading description of study sites which was addressed by the author checking the nature of the sites on detailed aerial photographs. It was noted that "deep peat dominated by vigorous heather is usually a sign the land management is causing the peat to dry out. Burning of tall heather makes growth more vigorous, leading to further drying out and oxidation of the catotelm peat by healthy vascular-plant root systems. Tall heather can additionally encourage hot fires which further damage or destroy the ground layer of *Sphagnum* mosses. In contrast, heather which grows within a vigorous *Sphagnum* carpet does not enter the classic 'growth and collapse' cycle which drives demand for a cycle of burning management. Heather shoots growing in a *Sphagnum* carpet are continually engulfed by the vigorously-growing *Sphagnum,* causing the heather to send out fresh shoots and roots within the *Sphagnum* layer (heather 'layering') and thereby heather growth remains relatively young and vigorous." "The carbon balance of burned peatlands generally involves a loss of carbon from the peat store as well as loss of peat-forming capability. Natural fire frequencies on *Sphagnum* peat bogs in boreal Canada have been found to average 1,150 years (Kuhry 1994). It has been shown that if the fire frequency is 5-7x greater than this average (ie between 164-230 years) then the resulting long-term losses from the carbon store mean that the bog achieves zero carbon sequestration. If the fire frequency is more than this, the bog goes into long-term carbon deficit. The average fire frequency in the past 100 years or so on British blanket mire has been approximately 30 years." "Studies indicating that there may be short-term carbon gains from certain types of burning on peatlands in Britain need careful interpretation, particularly in terms of the specific nature of the peatland under study. Such work has not fully addressed the complete carbon balance nor the long-term implications of burning."

Other review on carbon balance

Gray & Levy (2009) [4+] reviewed carbon flux research in UK peatlands in relation to fire and the Cairngorms National Park (CNP). In addition to studies from Moor House, eight papers reporting fluxes of CO_2 and nine of methane were examined. However, few of the studies related directly to managed burning, although Hogg *et al.* (1992) "found that burning increased methane fluxes in peat cores, in agreement with immediate post-fire methane flux data from … Caithness and Sutherland" (Gray 2006). The authors developed a simple model of peatland carbon balance as affected by fire (parameterised with data from Ward *et al.* 2007 [1+], above) to illustrate the potential for modelling and to analyse some of the key sensitivities. The results showed a peak in carbon storage with a fire period around 30 year, with a sharp decline in carbon storage when fire period decreased below 15 year, though the authors noted that the results were illustrative rather than predictive mostly due to data deficiencies. They concluded that "there is insufficient data at present for firm conclusions to be drawn on the effects of fire on the carbon cycle of CNP peatlands or to sufficiently parameterise local or national models. Much of the evidence is from a single site (Moor House in the Pennines) and whether the patterns found there are applicable within the CNP is open to question." It was also noted that there are "significant peat deposits with high carbon content of a lesser depth than is usually used to classify peatland" and that "these may be more at risk from wildfire than the deeper,

wetter deposits. In terms of assessing fire effects on carbon cycle processes in peatlands, we agree with Davies *et al.* (2008) that that removing the variable depth distinction to define a peatland would give a more holistic approach to soil carbon; a classification of soils based on carbon content would be helpful."

Appendix 7 The studies and evidence for the effects of burning on water quality and flow

Twenty-one+ studies all from the England (20) and Scotland (2 overlapping with England) provide evidence on the effects of burning on water quality and flow. In addition, the findings of the main reviews that relate to the sub-question are also summarised separately below.

Hard Hill plots hydrology and water chemistry studies

Hard Hill plots hydrology and water chemistry/quality studies, 2005-2008 [1+] are described in a series of papers by Worrall *et al.* (2007), Worrall & Adamson (2008) and Clay *et al.* (2009a,b, 2010a). The studies took place in just two of the four blocks (A and B on the lower part of the slope) of the partially randomised block experiment involving two grazing and three burning treatments and involved the measurement of water table and sampling of water in three dipwells per treatment plot (x 12) for which various water chemistry parameters, including colouration and DOC aluminium, iron, calcium, sodium, magnesium, potassium, sulphate, chloride, bromide, fluoride, phosphate and nitrate concentrations, were determined. The published studies took place between April 2005 and January 2008, with Worrall *et al.* (2007) and Worrall & Adamson (2008) covering the initial period in summer 2005 towards the end of the burn period when both 10- and 20-year treatments were c.10 year since the previous burn (ie the only difference was in the length of previous rotations and hence the number of burns since 1954 – four for the 10-year and two for the 20-year treatment). Clay *et al.* (2009a, b, 2010a) cover an extended period (including summer 2005) before and after a burn in the 10-year treatments in February 2007 and include the incorporation of some additional measurements: hydraulic conductivity and runoff occurrence and water chemistry including colouration/DOC. The findings from the publications from this study are described below split by hydrology and water quality, and water chemistry (but as they are regarded as part of a single study are not given separate study type/quality classes).

Hydrology and water quality

Worrall *et al.* (2007) reported on the results of fortnightly sampling between April and September 2005 (16 summer sampling visits). Depth to water table was significantly different between both burning and grazing regimes; it was deepest in the unburnt/ungrazed plots and deeper under the 10-year than the 20-year treatment (though at the time they were both the same age since the last burn). There was no significant effect of grazing on pH, but it was significantly lower under burning although there was no difference between the two rotations. Conductivity was similarly significantly lower under burning and lowest under the 10-year treatment with grazing. DOC and absorbance (at 400 nm) showed similar patterns with a significant effect of days sampled and, to a lesser extent, burning (burnt lower than unburnt for DOC and lower for 10-year than 20-year treatments), but not grazing.

Clay *et al.* (2009a, b) extended the sampling of the same plots up to and after the burn of the 10-year plots in February 2007. Thus they report on the period April 2005 to January 2008 (52-59 all-year sampling visits over a 34 month period including all the data collected by Worrall *et al.* 2007), hence covering both the end of the burn cycle and immediately after the burn. Additionally, hydraulic conductivity was measured and runoff occurrence and quality recorded using three crest-fall traps per plot to intercept surface flow across the plots.

Clay *et al.* (2009a) reported on the extended results for water table, runoff and hydraulic conductivity. For the extended pre-burn period, unlike Worrall *et al.* (2007) who showed that it was the 10-year grazed plots that had the shallowest water table, it was the 20-year grazed plots that had the shallowest water table (35% c.f. unburned), though both were then the same age post-burn. Water

tables were significantly shallower after the burn though the effect size was relatively small (7% c.f. pre-burn). Hydraulic conductivity was significantly lower on the 20-year burn plots. Runoff occurred significantly more frequently on the recently burnt plots.

Clay *et al.* (2009b) reported on the extended results for water colouration and DOC. At the end of the burn cycle, DOC concentrations in soil water and runoff were not significantly affected by burning treatment (which might be expected as both the 10- and 20-year rotations were, at that stage, 10 year after the last burn). These findings were contrary to those of Worrall *et al.* (2007) over a shorter period, which showed lower DOC for burnt than unburnt and for 10-year than 20-year treatments. However, colouration (absorbance at 400 nm) was significantly lower on 20-year burn plots than the unburnt controls. In the weeks following the burn, there were peaks in DOC concentration and colour in the soil water (between three and seven weeks after burning) compared to unburnt controls but these peaks were relatively short-lived. However, Composition of DOC in soil and runoff water was not affected by burning treatment; rather, the variation was controlled by time of sampling and season, with month explaining up to 23% of the variation in soil water. Values for DOC, colour and derived specific absorbance were significantly lower in runoff water than soil water. Grazing did not significantly affect these parameters in soil water and the end of the burn cycle, though there were significant effects in runoff water at the end of the 10-year cycle.

Water chemistry

Worrall & Adamson (2008) reported on the results of fortnightly sampling between April and September 2005 (16 summer sampling visits).showed that calcium, sodium, magnesium and phosphate concentrations were significantly lower on all burnt plots, with only aluminium concentration being significantly higher on burnt plots. Only chloride showed a significant change (decrease) with the presence of sheep grazing (and only when plots were also burnt). A principal components analysis showed that the composition of most soil waters studied could be described by rainwater and soil water components, but in unburnt plots, a base-rich, high ionic strength water was sometimes present. The study suggested that burning, but not grazing, caused significant changes in soil water composition. However, no evidence was found for structural change in the soils even after long term (50+ years) grazing and burning management.

Clay *et al.* (2010a) extended the sampling of the same plots up to and after the burn of the 10-year plots in February 2007, thus covering the period April 2005 to January 2008 (59 all-year sampling visits over a 34 month period including all the data collected by Worrall & Adamson (2008), hence covering both the end of the burn cycle and immediately after the burn. The results showed differences in concentrations between chemical species and water types. After the burn of the 10-year treatment plots, aluminium, iron and sodium showed significant increases and calcium, chlorine and bromine significant declines in soil water. Aluminium, calcium iron and NO_3 showed significant decreases following burning, while the largest change was a nearly five-fold increase in calcium, in runoff water. Principle components analysis indicated three different water sources that make up soil water compositions on the site: base-rich groundwater similar to as reported by Worrall & Adamson (2008); a shallow water component; and a component with high loadings for chlorine and SO_4 suggesting a rainwater composition. Burning resulted in lower soil water concentrations of those species associated with groundwater, for example, calcium and magnesium and dilution of runoff water.

Ward *et al.* (2007) [1+] (described under sub-question c) did not find any significant difference in DOC in soil water between the unburned and 10-year burn plots towards the end of the burn cycle which is consistent with the findings of Clay *et al.* (2009b), but contra Worrall *et al.* (2007).

Water colouration/DOC

The evidence on water colouration/DOC is organised below by scale, from laboratory studies to plot-based studies, catchment-scale studies and modelling. Some of the studies cover other aspects of hydrology and water chemistry but the findings are reported together in this section.

Laboratory studies

Allen (1964) [2++] conducted a multi-faceted mainly laboratory investigation into what happens to *Calluna* when it is burnt, which nutrients are released and how they are filtered and retained by different soil types. He measured: K, Ca, Mg, P, in *Calluna* ash, comparing differences dissolved in pure water or rain water; K, Ca, Mg, P and N in *Calluna* ash created at 500°C and 900°C; K, Ca and P in fresh *Calluna*, partially decomposed litter and fully decomposed litter; the rate of leaching ml/h through different soil types both burnt and unburnt; the amount of K, Ca, Mg and P left in leachate after moving through different soils, again both burnt and unburnt; the amount of extractable nutrients K, Ca, Mg, P, NH4 and NO3 at different depths of soil, both burnt and unburnt; quantities of K, Ca, Mg and P retained by peat, clay, sandstone and limestone soils when treated with simulated ash extract; and the amount of K, Ca, Mg retained by fresh, heated and dead *Sphagnum* .

In summary, mineral nutrients, particularly K, were readily dissolved from ash from burnt *Calluna*. The rate of solution was reduced if *Calluna* was burnt at a higher temperature. Soils tend to retain dissolved nutrients as rainwater leaches through, with organic and clay soils being more efficient than sandy soils. *Sphagnum* also retains dissolved nutrients. Over half the carbon, nitrogen and sulphur in heather was driven off in smoke. However, any losses from the system can be restored from rainfall within a relatively short period except on porous soils. Nitrogen may take longer, but microbial action might be important for this. Burning did not appear to make much difference to the way nutrients were filtered through the soil. Fine ash washed into the top surface seemed to slow the movement of water. Tests on soils showed that nutrients were held by litter and upper peat layers and, again, there appeared to be no direct impact of burning on this. This appeared to be true for mineral soils with a thin organic layer and for deeper peats. Nutrients were held in the upper few centimetres of soil. Importantly, burning raised the pH of soils both initially and after leaching with mildly acidic water. This may be important as it has implications for DOC production since pH controls the solubility of DOC (Thurman 1985), with the higher the Ph the greater the solubility of DOC. *Sphagnum* was very efficient at retaining nutrients dissolved in water, with heated (up to 60°C) *Sphagnum* being nearly as efficient as fresh *Sphagnum* . Dead *Sphagnum* was less efficient but did still hold nutrients. Less than 1% of mineral elements were lost in smoke, although burn temperature did have an impact with higher temperatures causing greater losses. However, more volatile compounds carbon, nitrogen and sulphur are lost, again with greater losses at higher temperatures. Approx 70% of nitrogen is driven off and 50% of sulphur. 60.5% of carbon is lost in smoke from burning heather at 550-650°C and 67.5% of carbon is lost in smoke if burnt at 800-825°C.

Miller (2008) [2++] ran experiments using cores from a burnt area with bare peat, a burnt area where vegetation was regrowing, an unburnt bare peat and an unburnt vegetated peat. Some of the cores from each treatment were inoculated with microbes and some were subject to different drying and wetting cycles. Overall, increased water colour release in leachate was detected from burnt peat compared with unburnt peat. Drying and re-wetting of burnt peat had little effect on colour or DOC release. There was a significant drying and inoculation interaction, with enhanced colour release for cores that had been dried and inoculated, immediately after the burn. One year after the burn, this drying/inoculation interaction had been lost in the bare peat but not in the burnt revegetated peat. This difference between the years for the bare peat was suggested to be due to soil moisture content differences affecting the soil microbial community but it was noted that further research was needed to clarify this. Miller (2008) also noted that it was very clear from the experiment that the effects of burning on water colour release lasted more than one year. However, this study investigated an accidental burn which occurred during the summer time, and the results, therefore, are unlikely to be applicable to burning conducted over winter as part of grouse moor management.

McDonald *et al.* (1991) [2++] provided leachate data from peat cores. They showed that one month after burning there was no difference in colour leaching from burnt and unburnt peat cores. However, over longer periods prescribed burning increased colour in leachate compared to unburnt peat cores vegetated with *Calluna* and Eriophorum. Some of the peat cores were burnt in the field by normal planned controlled fires and then extracted and returned to the laboratory whereas other peat cores has their vegetation burnt at controlled hot and cooler temperatures in the laboratory and were then leached with water. Hotter burns were associated with more colour release than cooler burns. This

was the case for both the controlled laboratory burns and for field samples. However, in the field 'hot burns' were assumed from the state of the peat surface and vegetation rather than by temperature measurement or control. The lag between burning and colour increase was used as evidence to suggest that burning did not directly cause colour increases but lead to other changes which then in turn lead to colour increases. It was suggested that these processes might include accelerated microbial decomposition in warmer temperatures below an unvegetated peat surface compared to under a cooler vegetated surface (see also below regarding related catchment-scale studies.)

Plot-scale studies

Water colouration/DOC aspects of the studies by Worrall *et al.* (2007) and Clay *et al.* (2009b) at the Hard Hill experimental burning plots are dealt with above and are not repeated here.

Worrall *et al.* (2012) [2-] studied the effects of burning and cutting *Calluna*-dominated vegetation on deep peat on DOC in the Goyt Valley in the Peak District. They measured soil water and surface runoff under a range of treatments including presumed management burning and cutting with or without removal of cut material. Soil water was sampled in dipwells and surface water by traps. Eight treatment combinations were investigated plus a control.

Results were combined so that reporting focussed on differences between cut, burned and control treatments. Few significant differences were detected between treatments and a high proportion of variance was not explained by treatment. Depth to water table was reduced by cutting and burning relative to controls, possibly due to changes in evapotranspiration, though results for new burns were not significantly different from the control. There was no significant difference between removing or leaving material after cutting. Differences in surface water runoff between treatments were not significant, but soil DOC decreased under cutting and burning treatments. It was suggested that this may have resulted from reduced depth to water table and/or dilution of soil water by surface water or rain water which is low in DOC. These studies were conducted on relatively 'dry' peat (average water table depth 42 cm) and it was suggested that sites where water tables are normally closer to the surface may see little effect. This study did not consider whether flows are affected by hydraulic conductivity or macropores.

Clay *et al.* (2012) [2+] studied the colour and DOC characteristics of soil and runoff water in a single chronosequence of post-burn ages to ten years from two sites in the Northumberland uplands managed for grouse. The vegetation appears to have been *Calluna-Eriophorum* blanket mire mostly over deep peat. Absorbance (at 400, 465 and 665 nm), pH, conductivity and DOC concentration were determined from water samples taken in consecutive months (bar one month of extreme weather) over a 15-month period from dipwells and crest-fall runoff traps in eight burn ages of different ages plus three unburnt control plots at each of the two sites. Colour: carbon ratio and specific absorbance (extent of humification) were derived from these. Water table depth was measured from dipwells.

General linear modelling of the blocked ANOVA design, with and without covariates, was used to explore sources of variation in the data, with pairwise comparisons among factor levels investigated using the Tukey test. Analyses were performed on both standard data sets (log transformed were appropriate, as indicated by the Levene test) and those normalised using the average of six control values; this allowed for seasonal changes to be absorbed. The results showed spatial and temporal variations in many variables. It was suggested that basic colour (Abs 400) and DOC concentration can behave differently within and between flow paths, so colour may not be an acceptable proxy for DOC. Although colour increased in the first four years of the post-burn period, DOC concentration did not show consistent trends. Run-off water was more dilute than soil water. It was suggested that trends in plot or catchment-scale DOC observed following burning, which do not correspond to the individual responses of runoff and soil water variables, may be mediated by changes in the balance of the two flow-paths, ie increased runoff in newly burnt areas. Extrapolating from plot-scale studies may be inappropriate because inconsistencies between these and catchment-scale behaviour are reported elsewhere.

Catchment-scale studies

Mitchell & McDonald (1995)/McDonald *et al.* (1991) [2-] studied factors involved in increases in water colouration based on their field observations in northern England, particularly in the Nidd and Washburn catchments in Yorkshire. McDonald *et al.* (1991) suggested that key factors contributing to increased colour risk in upland peatlands were: drought conditions, area of open-cut drainage, area of pre-afforestation ditching, areas of severely burnt moorland, south facing slopes, and areas of bare eroded peat. In their original report, McDonald *et al.* (1991) state that the links between burning and colour were not clear cut, mainly because the data on burnt area, types of burn and other confounding factors (for example, drainage and burning occurring together etc) were rather limited. Later, Mitchell and McDonald (1995) suggested there was a link between colour and burning but it was acknowledged that there was insufficient data for a statistical validation of this link.

O'Brien *et al.* (2005)/O'Brien 2009 [2-] attempted to clarify the relationship between water colour and burning regime in a correlation study within the catchment of the upper Derwent in the Peak District. Taking water samples at a number of points in sub-catchments within the upper catchment, they measured true water colour, pH, conductivity and altitude. From digitised aerial photos, they estimated the percentage of the area burnt within the catchment areas for the period 1999-2003 and 1999-2005.

Overall they found: no significant relationship between mean true water colour (2005-6) and percentage area burnt in either 1999-2003 or 1999-2005, although there was an indication of a weak positive relationship. One sub-catchment had high percentage of burning, but produced low mean water colour; it was suggested that other factors were responsible, for example, lower altitude and soil type (less peat). There was a significant negative relationship between water colour and pH ($p = 0.000$) and altitude ($p = 0.000$) but only a weak linear relationship between mean water colour and conductivity which was not statistically significant. A cluster of high water colour values were identified from high altitude sites on Winter Hill Peat and overall water colour was significantly higher on deep eat than acid loam soil ($p = 0.000$, mean 133 pH higher on peat), as would be expected. These results did not produce significant correlations between water colour and burning regime, perhaps partly reflecting the small number of replicates and lack of precision in recording the extent of recent burning.

Yallop *et al.* (2008)/White (2004) [2+] carried out a pilot correlation study on the influence of management burning on water colour. They looked at three catchments in Yorkshire that had good data from long-term water treatment works (WTW) for well-defined, unambiguous moorland catchments. Using aerial photographs from 2000 and 2005, they mapped the proportions of major vegetation types and burning. Four burn age classes were identified and soil types determined from the UK National Soil Resource Institute (NSRI) digital soil map. They studied 13 smaller, discrete sub-catchments within the 3 main catchments (with two-seven/main catchment). The water draining each was sampled in January 2001 with colour determined as Hazen units by the platinum-cobalt standard method using a spectrophotometer. In addition, three areas of differing vegetation type on deep peat were studied: new managed *Calluna* burn (<2 years old); an adjacent block of closed canopy regenerating *Calluna*; and an area of grass/sedge moor. All three sites were clustered within <10 m, therefore experiencing essentially identical hydrological and meteorological conditions. The percentage area of soil types, burn class and factorial combinations of these were tested as predictors for water colour sampled in watercourses draining each sub-catchment.

There were 13 combinations of soil, landcover and management class in the sub-catchments; only one, the area of exposed peat surface from recent burn management on deep peat soils was accepted into the regression (adjusted $r2 = 0.82$, $p < 0.0001$) showing it was the strongest predictor of water colour for the January water samples, all other factors were excluded with a strong linear relationship between increased burning and increased water colour in January. Similarly for the long term water colour monitoring, only the area of exposed peat surface from recent burn management for grouse on deep peat soils was accepted into the regression (adjusted $r2 = 0.94$, $p < 0.0001$). There was a similarly strong linear relationship between exposed peat from burning and average water colour for both 1992-1999 and 2002-2005. In the discussion there is a description of work done

on soil temperature and water table under different burning regimes/land cover. Soil temperatures under the exposed peat surface of new burn were on average 2-4 degrees warmer than the other two sites. Water tables were consistently high and near to the soil surface under grass/sedge moor. Both *Calluna* sites had considerably lower water tables. Water table oscillations were apparent under the area of new burn showing that burnt areas dry deeper in dry periods but rapidly rewet because of an absence of canopy during even short periods of rainfall. The discussion states that there is a highly significant relationship between the fraction of new management burns on deep peat and DOC release. Other research suggests that lowering of water tables and soil temperature increases have been implicated in enhanced release of carbon from the soil organic pool. The work suggests that soil temperatures are higher and the water table oscillates under recent burns. The results suggest that burnt areas dry deeper in dry periods but rapidly rewet because of an absence of canopy during even short periods of rainfall. Such cycles would ensure both high DOC production and rapid removal from the profile and delivery to drainage systems.

Beharry-Borg (2009) [2+] repeatedly surveyed 27 stream sites across the Upper Nidderdale region in Yorkshire over a 12-month period. This showed a significant positive relationship between the proportion of *Calluna* cover and DOC. The proportion of the catchment area burnt was associated with a change in the composition of DOC (reported as SUVA and also as a colour to DOC ratio). It was suggested that burning is associated with an effect on DOC.

Yallop & Clutterbuck (2009) [2++] examined the effect of land use and management on Dissolved Organic Carbon (DOC) production at two scales: within 50 discrete catchments <3 km^2 and at a larger scale in eight catchments where water colour data from water treatment works are available. The sites were in the South, West and North Yorkshire Pennines and on the North York Moors. Catchments were characterised into three types according to soils (blanket peat, peaty topsoils and non-peaty soils), vegetation character and presence of drains. Fire management was assessed from air photographs and extent of burns measured in four classes (‘new’, <c.4 year old; ‘recent’, c.4 -8 year; and two classes of ‘older’ burns, c.8-20 year and older). DOC was measured in outflow streams for the small catchments and from hazen colour data from water treatment works from the larger catchments.

The proportion of the catchment as blanket bog was highly correlated to DOC ($p < 0.001$). The most significant predictor of DOC was the proportion of new burn. The degree of variance explained increased when recent burning was added to the regression. The proportion of new burn on blanket bog was the most significant variable affecting DOC in the three Pennine areas but no relationships between catchment variables and DOC were found in the North York Moors sample area. No significant relationships were found between rainfall prior to sampling and DOC. For the eight larger catchments new burning explained 64% of the variance in DOC concentration between catchments. The results are interpreted to suggest that the management of vegetation on peat soils by burning is of relevance to water utilities as it affects water quality. The recent upward trend of colour in water may be due to recent changes in management. The lack of relationship between burning and DOC in other soil types (for example, at the North York Moors study area), suggests that it is not burning per se that affects DOC release but burning on deep peat soils. A suggested causal mechanism for the effect of burning on DOC release is through an increase in humic substances from peat decomposition. The changes in the hydrological and microbiological environment caused by exposure of the peat surface by burning could lead to increased decomposition. The mean surface area of exposed peat in new burns in this study is 84%. Bare peat is then exposed to increased solar radiation. Removal of the canopy also increases infiltration and through flow and this may also contribute to drying of the soil profile also resulting in an increase in microbial activity.

Chapman *et al.* (2010, 2011) [2-] compared the spatial and temporal variability of water colouration for fifteen watercourses in the How Stean catchment in Upper Nidderdale, in the Yorkshire Dales, in 1986 and 2006/7.

They observed that water colour increased in all sub-catchments between 1986 and 2006/07, but that there was considerable variability in the increase, which ranged from 22 to 155%. Although the study did not set out to investigate the effect of burning (Holden *et al.* 2012), six of the sub-

catchments were intensively managed by burning in both 1986 and 2006, five were not burnt over the twenty year period and four were not managed for grouse in 1986 but had very small (<4%) areas of burning occurring post-2000. Despite this variation in burn management, no relationship between burning management and increase in water colour was apparent. However, the method used to determine the extent of burning and the fact that it did not separate out recent burning was criticised by Yallop *et al.* (2011). For the catchments that were not managed by burning over the 20-year period, water colour increased between 22 and 117%, whereas for the catchments that were consistently managed by burning, water colour increased by 37-123%. Hence both types of catchments displayed a wide variation in the increase in water colour over the 20 year suggesting that factors other than burning, such as interactions of decreases in sulphate deposition with different soil types were more important in controlling the variability in water colour increase in these catchments.

Clutterbuck & Yallop (2010)/Clutterbuck 2009 [2++] used historical data from five catchments in the central and Southern Pennines and one in the North Pennines to investigate factors affecting DOC production from upland catchments.

Four of the six catchments showed highly significant increases of 53-92% in DOC in the period 1990-2005. Large-scale changes such as changes in temperature and sulphur deposition explain 20-30% of this trend but a local factor, rapid expansion of moorland burning on blanket bog in the catchments, explained 80% of the increase. Smaller increases in DOC were found in two catchments where the changes in temperature and sulphur deposition occurred but where changes in burning were not apparent or of small magnitude. It was concluded that regional-scale factors underlie some of the observed change in DOC, but that changes in land management, ie extension of burning on to blanket peat, was a more important driver.

Yallop *et al.* (2010) [2++] investigated the influence of temperature, acid deposition and land use changes on the efflux of 'humic DOC' (hDOC) in three adjacent reservoir catchments in the South Pennines, for which there were long-term data from 1975 to 2005. The estimation of efflux requires monthly runoff values, which were estimated from a regression of monthly rainfall and runoff data from 29 catchments across the UK. Seven predictor sites with similar physical characteristics to the study sites, as determined by multivariate analysis, were used to validate the model. Water colour data were available as Hazen of UV absorption; these were converted to hDOC using bespoke empirical relationships or previously published convertions. Monthly efflux per catchment was estimated from hDOC concentration and runoff and expressed as annual fluvial efflux per m^2 of blanket peat, quantified from aerial imagery (since blanket peat is the primary source of hDOC in uplands). Changes in land management were computed using GIS by applying land and four burn scar age class categories (utilised by Yallop *et al.* 2005, 2006b) to successive aerial images for six discrete occasions between 1976 and 2005. Temperature and acid deposition data were obtained from the closest available monitoring stations. Missing values in the most comparable local temperature data set were circumvented by modifying local but urban data for that period using a regression of the relationship between both data sets. Seasonal Kendall tests were used to identify trends in hDOC export. Forward-entry multiple regressions were used to examine the relationship between annual hDOC flux and each of the explanatory variables as well as their interactions, for pooled catchment data initially; individual regressions were examined for those factors not selected. This was repeated for individual catchments, although with reduced statistical power. Data sets were first constrained to the period 1989-2005 and the strength of relationships compared to when 1976 data were included.

There was a strong concordance between hDOC efflux and concentration. Estimated fluvial export of hDOC increased over time, for example 9.4-11.8 g $m^{-2}yr^{-1}$ in 1976 and 27.7-29.3 g $m^{-2}yr^{-1}$ in 2000. The proportion of blanket peat within new burns was found to be the most significant explanatory variable for the two pooled and three individual data sets ($p = 0.009$, 0.001 and 0.05 for each site), with a strong positive relationship evident. There was a weak effect of acid deposition for the pooled data only. Newly burnt peat was responsible for a 5-15-fold increase in hDOC efflux over the estimated background level suggesting that net export of C is now occurring. Aerial imagery demonstrates a significant increase in the intensity of moorland burning over the previous four

decades. Figures for export of C from blanket peat in the Pennines were comparable in magnitude to those reported for other areas of the UK and elsewhere.

Armstrong *et al.* (2012) [3+] studied the relationship between surface vegetation and concentrations of DOC in soil water and in surface waters from drained blanket bog catchments. Soil water was sampled in dipwells sunk into peat in five types of vegetation (burnt, *Calluna*, *Molinia*, *Sphagnum* and sedges) on moorland managed for grouse and sheep grazing at Bingley Moor, Yorkshire. In addition, samples of surface water were collected from 32 drained blanket bog catchments in northern England and northern Scotland. In these, vegetation was allocated to three classes: *Calluna*-dominated, sedge-dominated or mixed. DOC concentrations were measured to investigate differences between vegetation types.

Calluna dominance was associated with highest DOC concentrations and both Molina and *Sphagnum* with lower concentrations. Samples from sedge vegetation were intermediate. Water from drained catchments with dominant *Calluna* had higher concentrations than in soil water samples from *Calluna* vegetation. The interactions between vegetation types and attributes of the physical and biotic environment were considered in an attempt to explain the differences in DOC shown. These factors need further study. It was concluded that it may be possible to manage vegetation to reduce DOC, carbon losses and water treatment costs. This might include actively spreading *Sphagnum.*

Modelling

Grayson *et al.* (2012, 2008) [2+] combined GIS with a multi-criteria evaluation model including spatial multi-attribute decision making based on simple additive weighting to predict the propensity of upland peatland catchments to generate water colour. Increased water colouration in upland streams over the last decade is considered to be primarily a reflection of increased DOC from the bacterial breakdown of organic matter in peat. Chemical treatment to reduce colour has significant associated human health risks because DOC interacts with chlorine to produce carcinogenic compounds. This study utilised long-term water colour data collected by the water industry for monitoring drinking water supplies at water treatment works. A subset of 18 catchments across the South Pennines and Peak District were selected to parameterise the model. Water colour data from each (collected by Yorkshire Water) could be related directly to a specific catchment, for which there are no other inputs of raw water, and were of a suitable resolution spanning 1995-2006. A suite of parameters were selected for which there is a proven relationship in the published literature to water colour production: soil type and superficial geology, rainfall, grip densities and topographic index (reflecting the effect of grips on local topography), natural hydrology, burn area, vegetation type and afforestation, slope, aspect and land tenancy (land owned by water companies was deemed less liable to generate colour). Aerial photographs (verified by ground-truthing) and national maps were used to quantify each of these across all 18 catchments. The model requires standardisation of all parameters to a common linear scale: the validity of this was assessed by comparing the significance for each parameter and catchment of the relationship between empirical data and mean water colour with that of the rescaled values and mean water colour. Parameters were then weighted according to the perceived ability to affect water colour, verified on the basis of statistically significant relationships within the data set. The model was accepted following a step-wise readjustment of the weightings to maximise the correlation coefficient for colour model score versus mean colour.

The final model explained c.90% of the variation in mean colour among the 18 catchments and allowed parameters with weak explanatory power to be removed, ie afforestation, precipitation, slope, aspect and tenancy. The key factors determining water discolouration were the extent of burning and the proportion of dwarf shrub (mostly *Calluna*) vegetation. The density of grips was only important where these were most prevalent. The model was validated by application to the upper Nidd catchment, North Yorkshire, for which the predicted and realised mean water colour was 96.6 and 96.4 mgl-1 PtCo, respectively. The authors suggested that the model could be used to identify water colouration hot spots within a single catchment, enabling cost-effective, strategically targeted changes to land use/management such as local cessation of rotational heather burning.

Critical synthesis on DOC/water colouration

Holden *et al.* (2012, 2011) [2++] carried out a critical synthesis of studies exploring the relationship between water colouration and/or DOC concentration and the managed burning of upland peatland. Both variables affect drinking water quality with significant cost implications for the water service industry in complying with statutory requirements, irrespective of health issues. The authors aimed to try to reconcile apparent contradictions among three scales of published study: laboratory-based studies using core samples removed from burnt sites in which percolates are analysed; hydrological studies within plots subject to experimental burning (but designed originally for ecological study); and catchment-scale studies based on water quality data collected primarily by the water service industry. The critical timeframe for detecting changes in both variables change is discussed. The ways in which water colour is determined was considered as was its relationship to DOC and the use of ratios to denote the relationship; any inter-study comparison must identify clearly the parameters quantified as well as their legitimacy. Furthermore, to elucidate the response of DOC to burning at the catchment scale, only the portion of the hydrological flux that contributes to stream flow should be sampled. The balance of empirical evidence suggested that water movement below the top 5-10 cm of the peat surface layer may be irrelevant.

Plot scale studies had failed to confirm that burning results in increased water colour/DOC. Two studies did not relate to peat soils and six of the eight investigations utilised the permanent experimental burning plots at Hard Hill, within Moor House NNR. Significant potential problems were identified by the authors and Lindsay (2010) in relation to this site which may have affected some of the results: there is potential hydraulic connectivity among the four replicate blocks because of their relative positions on a sloping hillside; regular trampling over many years in and around the plots may have altered soil structure and function; it is difficult to standardise small-scale controlled burning, which may not achieve the same fire characteristics as larger stands; bulk-sampling of water from excessive depths of peat (to 90 cm) may obfuscate the critical processes if these are indeed limited to the top 5-10 cm of the peat layer.

In general, catchment scale studies supported the proposition that increased burning over the last 10-15 years has prompted the recorded increases in water colour and DOC. However, Holden *et al.* suggested that if managed burning was the sole driver, DOC of the order 115-345mg l^{-1} m should be expected in water derived from what are only small burnt areas of c.2 ha or less (scaled to reflect catchment-scale burn intensity) in order to generate the concentrations of DOC monitored in catchment outflow. None of the studies presented reported figures of this magnitude. Holden *et al.* concluded that there was a lack of empirical mechanistic evidence to support and explain a simple linkage between burning and increased water colour/DOC. There is increasing evidence that the catchment vegetation, and perhaps *Calluna* specifically (the target of managed burning), may be an influential factor. The authors emphasised that improved understanding of the hydraulic linkages within catchments and the fate of DOC on its pathway to the stream is essential if the mechanism driving changes in water quality is to be understood.

Water quality in peatland watercourses

The above catchment-scale studies in watercourses are also relevant to this issue but are not repeated in this section.

Water chemistry

Armstrong *et al.* (2009) [3-] carried out a small-scale pilot correlation study to examine, and attempt to explain, factors controlling vegetation response to grip blocking on blanket bog which they suggested was the first of its kind. One similar pair of grips showing different vegetation responses and five dissimilar grips in relation to size, vegetation and burning were surveyed on gripped blanket bog at Allenheads in the North Pennines. One of the seven grips had been burnt-over. The extent and composition of the revegetation, and grip morphology and water geochemistry as predictor variables, were recorded post grip blocking.

The authors concluded that recent burning “notably influenced the geochemistry within the grip.” The lowest pH and highest conductivity, DOC and colour (at all absorbencies measured) were found in the burnt grip. There was a positive relationship between DOC concentration and (adjacent) slope for all grips, except for the grip that was recently burnt.

Worrall & Warburton (2009) [2+] surveyed blanket bog grips at five sites (sampled by c.600 grip ‘sections’) in the North Pennines to identify which carried the most flow, water colour and were the most eroded, and which blocking techniques were most effective. Evidence of burning was included as an explanatory variable.

Blocking caused a significant decrease in water colour which was greatest when blocking occurred in areas of managed burning. Water colour in grip sections was highest where there was evidence of burning and of *Calluna*.

Watercourse aquatic invertebrates

Ramchunder *et al.* (2009) [2+] proposed a schematic model to link key changes in upland peatland systems following rotational burning based on a review of the literature. The model represents potential impacts on the storm hydrograph, suspended sediment transport, stream physicochemistry (specifically electrical conductivity and pH) and the biota of moorland streams.

The authors proposed that burned catchments will have a greater proportion of land exposed to wind and water erosion, and induced hydrophobicity, meaning faster, flashier run-off and higher peak flows will be observed. Suspended sediment concentrations in the burned catchment may also be higher compared to an intact catchment. Hydrophobic soils in the burned catchment will result in lower infiltration rates, and therefore the initial increase in electrical conductivity may be moderated, and pH is unlikely to alter significantly. Burning is expected to increase the concentrations of suspended peat detritus in stream systems, and this will smother primary producers when deposited. The abundance of grazers in the stream system is likely to be reduced due to smothering of the substrata with sediment and a lower abundance of food sources. Higher concentrations of suspended peat detritus may result in a higher abundance of collector-filters. The greater deposition of fine organic sediment will reduce habitat heterogeneity and lack of prey items will have negative knock-on effects on invertebrate predators and fish.

Aspray (2012) [2++] studied macroinvertebrate communities and ecosystem functioning in peatland streams. The overarching aim was to improve understanding surrounding the impacts of stressors to peatland streams and to contextualise this research with an improved knowledge of the dynamics of intact peatland streams. This included assessing the impacts of two catchment-scale drivers of change in peatland habitats (rotational heather burning and erosion) on stream ecosystems, examining physicochemistry, macroinvertebrates and ecosystem functioning across fifteen streams and examining gradients of sedimentation associated with environmental change and land management using streamside mesocosm and reach experiments. Fifteen study sites were located across the North Pennines, Yorkshire Dales and the Peak Distinct. Headwater streams were selected and catchments were classified as: (i) intact catchments, (ii) eroded catchments that are not actively managed, and (iii) catchments burnt by rotational heather burning.

Erosion, and to some degree rotational heather burning, were found to impact physicochemical variables, with total oxidised nitrogen (TON) and SSC displaying increased concentrations in impacted catchments. Associated shifts were found in macroinvertebrate communities, with amplified abundance in eroded catchments driven by increases in more sediment tolerate taxa, such as Chironomidae and Oligochaeta. Streams draining eroded and burnt catchments also displayed lower numbers of sensitive *Ephemeroptera, Plecoptera* and *Trichoptera taxa*. Functional parameters did not reflect these changes in chemistry and biota, but there were clear differences between the fifteen individual streams. It was concluded that “this body of research highlights peatland streams as unique and heterogenic systems but also as systems that are sensitive to anthropogenic stressors at both the catchment and reach scale. These habitats have intrinsic importance, supporting diverse macroinvertebrate communities, are significant for the modulation of carbon and are good indicators

of the condition of the surrounding catchment. This work emphasised the need for restorative measures and sustainable management in peatland habitats that considered the streams they support. In addition, the work furthers knowledge of the baseline conditions in these systems and increased understanding of the use of functional processes as ecological indicators in peatland streams."

Ramchunder *et al.* (2013)/Ramchunder (2010)/Brown *et al.* 2009 [2++] studied the effects of burning peatland catchments on benthic invertebrates in headwater streams in a sample of sites in the Pennines.

Brown *et al.* (2009) undertook a pilot study to compare the aquatic invertebrate communities of three second-order streams in upland peatland catchments subject to controlled burning with those of the Moor House NNR catchment in the North Pennines which has no burning and minimal grazing. Five quantitative samples were taken in September 2007 at each location. Data were pooled to provide estimates of total invertebrate abundance, taxonomic richness and the relative abundance of individual taxa, which were identified to species-level as far as possible. Identification was validated externally. Results were presented as a series of bar charts with standard errors. There is preliminary evidence that although burning does not appear to affect the total abundance of invertebrates or their taxonomic richness but it may be detrimental to some species/groups; this was exemplified by the scarcity of the mayfly *Ecdyonurus dispar* and the stoneflies *Isoperlodes grammatica* and *Perlodes microcephala* in streams from catchments with controlled burning.

Ramchunder *et al.* (2013) sampled a selection of three burned and three unburned sites over four seasons and five further examples of each were sampled on a single occasion. At each site macroinvertebrates were sampled using a Surber sampler and identified to species or higher taxonomic group (for example, Diptera to genus, Oligochaetes to class). Sixteen environmental factors were measured. Species data were used to calculate measures of community structure (total abundance, relative abundance of functional feeding groups, taxonomic richness, diversity index and taxonomic dominance (Berger Parker Index). Data were analysed show differences in environmental factors and biota between burned and unburned sites and to explore the species – habitat relationships. Both 3 x 3 (seasonal) and 5 x 5 (single sample) surveys showed burning was linked to changes in stream environmental variables (for example, increases in suspended sediment concentration, Fine particulate organic matter, Al, SO^4, NO^3, DOC and smaller D_{50}). This suggests that burning can increase the vulnerability of soil to physical erosion resulting in higher sediment yields. Significantly higher concentrations of DOC were found in burned catchments and this study suggests that burning is a local driving factor in DOC production. There were significant differences in community richness, diversity and dominance and community composition in contrast to artificially drained catchments where drainage was shown to have no such effect. There were lower abundances of herbivores and predators in burned sites and there was a shift from communities dominated by mayflies and large predatory stoneflies to communities dominated by Diptera (especially Chironomids and Simulids) and smaller stoneflies at burned sites. Higher amounts of Suspended Sediment Concentration, Fine Particulate Organic Matter in burned sites were considered to be likely to drive these effects. The generality of these results is difficult to determine as there are few other published studies into stream responses to heather burning, but the findings here are similar to those found from studies of the effects of wildfire in other locations.

Hydrology and water flow

Holden *et al.* (2013) [1+] studied infiltration, saturated hydraulic conductivity and macropore flow in deep peat on moorlands in the Pennines. Measurements were performed on: unburnt peat; where prescribed burning had taken place 2, 4 (both 'recent burn' treatments) and >15 years prior to sampling; and where a wildfire had taken place four months prior to sampling. Important flowpaths in the upper layers of blanket peat and were investigated through the use of tension disk infiltrometers.

Where there had been recent burning (<2yr, <4yr and wildfire site) saturated hydraulic conductivity was approximately three times lower than where there was no burning or where burning was last conducted >15 years ago. The contribution of macropore flow to overall infiltration was significantly

lower (between 12 and 25 % less) in the recently burnt treatments. There were no significant differences in saturated hydraulic conductivity or macropore flow between peat which had been subject to recent wildfire and those which had undergone recent prescribed burning (<2 and <4 yr). The results suggest fire influences the near-surface hydrological functioning of peatlands but that recovery in terms of saturated hydraulic conductivity and macropore flow may be possible within two decades if there are no further fires. Fire reduces the role of micropores and therefore increases the role of micropores in infiltration and percolation of water and may therefore increase the potential for leaching of dissolved organic carbon because there would be increased contact time between the water and peat. Possible mechanisms for the effect of burning on macropores are the production of fine sediment and ash which block macropores.

Review on hydrology and water flow

Pattison & Lane (2011) [4+] explored the problems associated with extrapolation to the catchment scale from small-scale or sub-catchment studies in which land-use management is observed to affect local hydrology, especially if such data are then used to formulate catchment or flood risk management policies.

They identify the various components of catchment hydrology and explore in depth from existing literature four examples to illustrate how flood response to land-use is not only highly site specific but also inextricably linked to climatic patterns: (i) the apparent causal relationship between increasing arable agriculture and exacerbated flood risk from runoff, which cannot be dissociated from long-term trends in flood risk associated with climate change; (ii) the complexity of the processes linking recent changes in pastoral agriculture to downstream flood risk; (iii) the contextual specificity of the impact of grip creation and blocking on the magnitude, duration and timing of flood peak flow; and (iv) the importance of spatial location in determining the efficacy and appropriateness of flood relief measures characteristics and the role of flood plain characteristics in alleviating peak flow and duration. They concluded that dividing a catchment into smaller units, for ease of, for example, scientific study or demonstration projects, causes a potentially unacceptable loss of explanatory power (and may lead to inappropriate catchment-scale flood risk management) and that the converse, simplistic process of linking units without regard to complex causal processes ignores the critical importance of scale, event and catchment specificity.

Appendix 8 The studies and evidence for the impact of differing characteristics of burns

Studies on the characteristics and behaviour of fires and post-burn effects

Davies (2005) [2+], in his thesis, described various investigations and reviews of both the behaviour and impact/severity of management fires "in recognition of the need to develop multi-aim land management practices that ensure both continued productivity and protection of biodiversity in the face of climatic and environmental change."

The author noted that "fuel structure and loading are crucial controlling factors on both fire behaviour and impact governing both rate of spread and heat release to the ground surface". Monitoring showed moisture content to be relatively stable temporally, but spatially variable. Periods of extreme low moisture content in early spring are associated with frozen ground, winter cuticle damage and physiological drought. Such conditions may have contributed to the large number of wildfires in spring 2003. The amount of water held in fuel particles is of vital importance to fire behaviour."

"The fuel moisture content (FMC) of both live and dead fuel components affects the duration of preheating and amount of energy required to raise fuel to combustion temperature (Pyne *et al.* 1996), rates of combustion and the amount of energy released (Byram 1959). High moisture content reduces fire risk, intensity and spread as energy produced by a fire must first be used to drive off excess moisture while the production of large quantities of steam may have a smothering affect on the combustion process by reducing the oxygen concentration of surrounding air (Catchpole & Catchpole 1991)." Increased temperatures and durations above 400°C in the canopy layer were correlated with higher fuel loads, lower humidity and fuel moisture and were also influenced by greater quantities of dead fuels. Although increases in below ground temperatures were never more than a few degrees these are nevertheless greater where fires are slower moving and hence less intense but have greater residence times."

"Vegetation height is important as it is not only strongly correlated with total and fine fuel loading but the height of the canopy above the ground may govern the degree of aeration of the fuel and thus oxygen supply for combustion. Old stands with heterogeneous canopies also have greater surface roughness which leads to turbulent air flow, greater penetration of air movement into the canopy and thus drying of fuels as well as an improved supply of oxygen for combustion. In older canopies there was, therefore, not just a continuous oxygen supply but also significantly less heat and time was needed to evaporate water before pyrolysis and combustion could begin. Fires in such conditions were more akin to miniature crown fires rather than the traditional perception of a surface fire which spreads through dead litter on the ground surface."

"Fires in older fuels burn faster and hotter than those in younger fuels and are more prone to rapid changes in behaviour associated with changing weather conditions. Several authors (Cheney *et al.* 1993, Fernandes 2001) as well as Hobbs and Gimingham (1984) have reported that fires spread faster with higher intensities where fire widths are greater and should therefore consider whether current blanket advice of 30-m wide fires (based on habitat usage by red grouse) is appropriate or whether when burning in higher fuel loads it is advisable to burn smaller fires."

Davies *et al.* (2010) [2+] investigated the nature of fuel moisture content (FMC) within the various strata and fuel components of the *Calluna* canopy, as well as its spatial and temporal variability as a means of understanding fire behaviour in managed burns and wildfires, particularly in relation to

extreme environmental conditions in spring. *Calluna* fuel components were categorised by vertical stratification, stem diameter and living versus dead biomass. All these components, plus the ground layer of mosses etc , were sampled by collecting biomass within quadrats located at random among upland moorland (NVC types H10, H12 and H16 all on peat to 30+ cm) in one or more of five geographically distinct locations in the Scottish uplands. The study described three separate investigations: small scale temporal and spatial variation, ie among strata and throughout consecutive spring and summer periods; daily sampling of strata throughout a four week period in autumn and an eight-week period the following spring supplemented by a six-day non-consecutive period at an additional site in spring; the effect on fuel components of exposure after extreme weather events in early spring.

The results were analysed by random and mixed effects linear models with any constraints resulting from unbalanced design recognised explicitly and comparisons limited accordingly. Significant temporal and spatial variation in FMC was observed, with rapid fluctuations coinciding with periods of dry weather, low temperatures and frozen ground. Small-scale within-site differences may be sufficient to affect the efficacy and/or predictability of managed burns. Physiological drought prompted by cold clear weather and frozen ground reduces significantly the FMC of live biomass and creates the potential for extreme fire behaviour, for example, in spring.

Davies & Legg (2011) [2+] studied the role of moisture content in components of *Calluna* moorland vegetation in determining fire risk. They conducted 20 ignition experiments at two *Calluna*-dominated moorland sites in Scotland (probably not peatland but soil/habitat type not given). Attempts were made to start fires using point or line ignition sources during the legal burning season in plots in which fuel load and structure had been measured. Fires were in late-building phase *Calluna* stands on flat ground where there was a mat of pleurocarpous mosses and with low cover of other species.

Data were given on the fuel moisture content of different components of the vegetation (generally correlated except that moisture content of dead material was not correlated with that of other components). Fuel moisture content of the upper vegetation varied from 29.7% to 163% of dry weight. The lower canopy had lower moisture content. The greatest variability of moisture content was in the moss layer (25% to 400%). Moss and litter did not burn where the moisture content exceeded 140% but at moisture content less than 70% significant amounts of smouldering continued after flaming combustion was extinguished. Data were analysed to model the probability of sustaining fires.

Thresholds for ignition and sustaining fire are strongly related to moisture content. Fires are unlikely to start where moisture content of the lower canopy exceeds 75% but fires may burn in the *Calluna* canopy independently of the litter layer. The role of dead matter in providing fuel needs further study. Once a fire has started the rate of spread was influenced mainly by the moisture content of dead foliage in the lower canopy and in the moss/litter layer. The study contributes to the understanding of the role of fuel moisture and informs the improvement of assessment of moorland fire hazard.

Benscoter *et al.* (2011) [1+] experimentally altered soil moisture profiles of peat monoliths and used laboratory burn tests to examine the effects of fuel type and depth-dependent variation in bulk density and moisture on depth of fuel consumption.

Results showed that mean depth of burn varied across the three moisture treatments ($p = 0.003$), with the air-dried and oven-dried samples burning to a greater depth than the field sample. Depth of burn was not significantly different ($p = 0.05$) among fuel types. Ignition at the soil surface showed no significant difference ($p = 0.05$) in bulk density between successful and unsuccessful ignitions. Average surface volumetric water content for successful ignitions was significantly less than for unburnt samples ($p = 0.03$).

Harris *et al.* (2011a) [2+] considered fire characteristics and vegetation biomass reduction in 17 (‘pressurised fuel assisted’) ‘cool’ management burns (in March and April) on *Calluna*-dominated vegetation on deep peat at Howden Moor in the Peak District. Fire temperature was measured at the vegetation base and canopy at four sample points in each burn and biomass reduction was measured through a comparison of pre-and post burn data.

The results showed that pre-burn mean dry biomass was 301 g. Maximum temperatures recorded were 982°C and 993°C at base and canopy, respectively. Burn residence time ranged from 11-547 seconds at the base and 6-474 seconds at the canopy. Combustion was incomplete in all burns. Up to 93% of total dry biomass was removed (mean 66%, range 21-93%) and up to 96% and 97% of litter and *Calluna* dry biomass, respectively. Thus, there was considerable variation in fire characteristics and biomass measurements, by 1-2 orders of magnitude, which was greater within than between burns.

Reviews

Legg & Davies (2009) [4+] reviewed what determines fire occurrence, fire behaviour and fire effects in heathlands in which they also consider peatlands. They note that "there are many different types of vegetation fire: natural fires and man-made fires; well-controlled management fires and wildfires; mild fires and severe fires; 'cool', slow-moving fires and hot, intense fires." "The combination of the frequency with which fires burn, how hot or intense they are and how damaging or severe they are is termed a "fire regime". Differences in fire regime are linked to climate, vegetation type and the degree of intervention." Regarding fuel, "peat and organic soils represent a completely different class of heathland fuel (from vegetation). Peat fires are smouldering fires, rather than flaming combustion. They are difficult to ignite but, once established, are very difficult to extinguish. Experimental evidence suggested that peat must have a moisture content below about 120% dry weight to support smouldering combustion (Rein *et al.* 2008). This suggests that peat fires are most likely to occur in summer and early autumn after periods of prolonged drought. The role of peat type or structure on fire is not yet understood, but piping and cracking of peat are likely to be associated with more rapid drying of the peat and hence an increase in the frequency of periods when peat fires are a possibility."

Whilst the rate of spread and 'fireline intensity' (rate of heat output per unit length of fire-front) are the two main characteristics of fire behaviour, the authors suggested that "they are poor indicators of damage caused by the fire (Hartford & Frandsen 1992, Neary *et al.* 1999)" and of more importance are the "residence time and depth of penetration into the soil of lethal temperatures." "These are much less well understood, but slow moving fires ... may have longer residence times than fast-moving fires that spread rapidly over the surface of the vegetation but have rather superficial effects (Ascoli *et al.* 2006). However, residence time depends less on wind speed and direction than on the nature of the fuel: coarse fuels will burn much more slowly than fine fuels, whatever the wind speed. Depth of penetration of heat in UK situations appears to depend on the moisture content of the litter and soil. On damp organic soils the temperature rarely rises more than a few degrees centigrade even 2 cm below the surface of compacted litter (Davies 2005). Where the litter is dry and can ignite, however, smouldering fires can establish. Although spreading at only a few centimetres per hour with temperatures much lower than those of flaming combustion, smouldering fires will consume large quantities of organic material, destroy seed banks and kill the rootstocks of plants (Maltby *et al.* 1990, Neary *et al.* 1999)."

The authors suggested that the effects, or severity, and recovery from the disturbance vary depending on the type of ecosystem and its condition including the extent to which it has been subject to, and modified by, previous fires. They suggested that the effects could be considered as short-, medium- and long-term. In the short-term, the immediate effect is destruction of the above-ground biomass and heating of the surface soil. The severity of the effects depend on the proportion of the fuel consumed and the depth and duration of the heat penetration below ground. Recovery depends on the nature of the vegetation and can be rapid, for example, a spring fire in *Molinia* litter will have very little effect on the regrowth of the dominant species. Similarly, normal burns in building phase *Calluna* will damage neither the seed bank nor rootstocks which will re-sprout rapidly, whereas fires after a prolonged dry spell that ignite the litter and peat may both kill the rootstocks and damage the seed bank from which recovery will be patchy. In the medium-term, fire can stimulate successional changes. When the majority of plant species survive a fire, the species composition may change little other than in abundance. However, the exposure of bare peat may allow new species to establish. "One consequence of regularly repeated fire is the dominance of single competitive species that suppress other species so deducing the local (alpha) diversity. Thus,

Calluna will come to dominate in on drier areas and *Molinia* or *Eriophorum vaginatum* on wetter sites." *Molinia* or *E. vaginatum* may become dominate very quickly, whereas *Calluna* takes longer and declines in the degenerate stage allowing other species suppressed in the dominant building/mature stages to increase. Thus, many small fires create a mosaic of different ages and diversity in species composition and structure increasing landscape-scale (beta) diversity, at least in heaths. "The long-term effects of fire, ie the effects that accumulate over several fire cycles, are less well understood."

Appendix 9 The studies and evidence for the interaction of burning and grazing

The main experimental study was again the Hard Hill burning and grazing experiment at Moor House NNR, with the results from the main vegetation studies (1961-2001), *Rubus chamaemorus* studies (1969), hydrology and water chemistry studies (2005-2008) and carbon dynamics (2003-2004) all considered.

The studies and their findings are described below ordered within sub categories by date of publication. In addition, the findings of two reviews that relate to the sub-question are also summarised separately.

Hard Hill burning and grazing experiment plots

Hard Hill burning and grazing experiment vegetation studies 1961-2001 [1++] at Moor House NNR in the North Pennies is described in detail earlier under sub-question a. Large grazing exclosures were established in 1954 in a partially randomised block design of four blocks at differing altitudes up a slope x 2 grazing treatments (grazed, ungrazed) x 3 burning treatments (no burning and 10-year and 20-year rotations, with burning only randomised within a grazing treatment for practicality) in 30 m x 30 m plots. The wider area was grazed by sheep at a low overall rate of 0.1/ha with no winter grazing, with the sheep having access to the unfenced plots.

Adamson & Kahl (2003) in an overview of the long-term effects of burning and grazing in the Hard Hill plots up to 2001 noted that “light grazing was shown to have very little effect on vegetation composition” which was attributed to the absence of stock during winter when dwarf shrubs are targeted preferentially. “*Rubus chamaemorus* was favoured by no grazing and burning attributed to highly selective grazing by sheep and improved light intensities in post-burn plots, respectively.” Lee at al. (2013) in their recent re-analysis of the data between 1972-2001 (18-47 year after the initial burn) found that “very light sheep grazing had little impact” and reported no significant grazing effects or interactions with burning. Some effects of grazing were reported in earlier surveys (when grazing levels may have been greater) and monitoring of the initial post-burn period (1-5 year in both burn treatments) between 1976-80. Rawes & Hobbs (1979) noted that in 1961, seven years after the initial burn, bare ground was more frequent in the grazed plots and that in 1972, 18 year after the initial burn, differences in species (for example, a decline in *Calluna*) between the burn treatments were greater in the grazed plots. Nevertheless, at this lightly summer-grazed site, the major changes in vegetation were related to burning rather than grazing.

Taylor & Marks (1971)/Marks & Taylor (1972) [1+] used one of the four replicate blocks at Hard Hill to separately study the effect of burning and grazing on the growth of Cloudberry *Rubus chamaemorus* (described further under sub-question a). This included an experiment to discriminate the effects of canopy removal and nutrient addition following burning, twenty x 1 m^2 samples within the 20 year burn rotation plots were clipped of all biomass down to the bog surface in April 1969. Bulked biomass was incinerated and the resulting ash was applied to 10 of the clipped plots, selected at random. The density of shoots and fruits produced in each 1 m^2 was monitored 3 and 15 months following treatment.

The studies showed that the density of shoots, total above ground biomass and rhizome biomass was highest in the 10 year burn rotation plots and much more so in the absence of grazing. However, individual shoot weights were similar between 10 year and 20 year burn rotations subject to grazing, where shoot density tended to drop between mid-July and August. Grazing caused a proliferation of smaller shoots with a lower propensity to flower and set fruit. There were no differences among

treatments in the concentrations in aerial dry matter of total N, P, K, Ca and Mg, so that total nutrient content per m^2 mirrored aerial biomass. The addition of inorganic nutrients to clipped vegetation was far less influential than grazing in constraining the density of shoots and flowers: plants in fenced plots accrued an even higher density of both in the second season after burning. The authors concluded that the ability of Cloudberry to maintain internal nutrient concentrations in plots where plants are released from grazing and able to proliferate vigorously demonstrates that nutrients do not limit its growth. Furthermore, sheep appear to seek out Cloudberry.

Hard Hill plots hydrology and water chemistry/quality studies, 2005-2008 [1+] are described in a series of papers by Worrall *et al.* (2007), Worrall & Adamson (2008) and Clay *et al.* (2009a,b, 2010a) (described further under sub-question d). However, there were few significant differences between grazing treatments or interactions between grazing and burning treatments. For the extended pre-burn period, the 20-year grazed plots had the shallowest water table (35% c.f. unburned).

Ward *et al.* (2007) [1++] sampled vegetation composition in their separate, wider study of carbon dynamics (described further under sub-question c) at the Hard Hills plots. This study only used the 10-year burn and unburned treatments for both grazed and ungrazed areas close to the end of the 10-year cycle.

Changes in the relative contribution of the three plant groups (graminoids, ericoids and bryophytes, although Sphagna were not separated) were observed due to burning and grazing, with the greatest effects being due to burning. Burning increased the biomass of graminoids by 88% relative to unburned controls ($p = 0.009$), but reduced the biomass of shrubs and bryophytes by 51% ($p < 0.0001$) and 92% ($p < 0.0001$), respectively. The change due to grazing was similar but of smaller magnitude. An 18% reduction in dwarf shrub biomass ($p = 0.05$) and a 47% reduction in bryophyte biomass ($p = 0.02$) was observed relative to ungrazed plots, but, perhaps surprisingly, there was no effect of grazing on the biomass of graminoids. No interaction between burning and grazing was reported. Burning and grazing reduced above ground C by 56% and 22% respectively and burning reduced C storage in the F and H horizons by 60%. Grazing did not significantly affect water colour or DOC in soil water at the end of the burn cycle, though there were significant effects in runoff water at the end of the 10-year cycle.

Burning and grazing interactions in other vegetation studies change

Miles (1971) [1+] investigated the effect of burning a mosaic of *Trichophorum-Eriophorum* blanket bog with abundant *Molinia* and *Molinia*-dominated grassland on grazing utilisation by red deer on the island of Rhum, Scotland.

Grazing was significantly greater in burned than unburned *Molinia*-dominated areas in the spring and summer following burning. Grazing rates declined again in both the two years after burning. Burning reduced litter by 80% on grassland and 85% on bog but within two years the quantity of litter had increased to half that on unburned areas. No changes in vascular plant cover were evident either due to burning or to excluding grazing. Burning killed most of the *Sphagnum* moss but this had mostly recovered after two years. New growth of *Molinia* on burned ground was attractive to grazing deer and burning therefore increased grazing value but such value was evident only in the year of burning and therefore increased the need to burn to maintain grazing value. The benefits of burning were short-lived. Alternatives to burning were considered; on *Molinia* grassland heavy grazing may encourage more palatable grasses such as *Agrostis* and *Festuca*. On *Molinia*-rich bog it was suggested that intensive grazing could prompt change to species-poor *Juncus squarrosus* bog. The study suggested that burning *Molinia*-rich vegetation is unlikely to result in benefits as it perpetuates *Molinia* dominance.

Rawes & Williams (1973)/Rawes & Hobbs (1979) [2+] described a separate unreplicated sheep-grazing trial carried out at Moor House NNR that included a burning followed by heavy grazing treatment (3.4 sheep/ha), light (0.37 sheep/ha) and heavy (3.4 sheep/ha) and an ungrazed control plot (but no burnt, ungrazed plot).

Rawes & Hobbs (1979) summarised the results after eight years. Under heavy grazing the initial botanical changes were similar in the burnt and unburnt plots: a rapid loss of *Calluna* and quick recovery of *Eriophorum vaginatum*. The cover of mosses was reduced in the heavy grazing plots, but liverworts showed a contradictory response as did *Rubus chamaemorus* probably reflecting differences in initial, patchy distributions. Lichens declined under heavy grazing and bare ground increased and after 7 year was at 11.4% and 8.6% in the burnt and unburnt heavy grazed plots. Heavy grazing with and without burning completely changed the structure of the vegetation to a shorter sward reflecting the loss of *Calluna*. Thus, in the initial stages there was little evidence that burning produced a difference compared to the unburnt, heavily grazed treatment.

Currall (1981)/Currall (1989) [2+] studied the effect of management burning on wet heath vegetation on the island of Skye in western Scotland in 1977-78 (described further under sub-question a) In addition, the effects of grazing, clipping and raking were investigated using grazing exclosures.

The author suggested that burning on short rotations and/or heavy grazing after burning can lead to maintenance of the dense graminoid phase (see under sub-question a) and a reduction in *Calluna*. Also that high grazing intensity and low burning frequency pushes the balance in favour of *Juncus squarrosus* and *Nardus stricta*, while the reverse pushes it in favour of *Molinia caerulea* and *Trichophorum cespitosum*.

Ross *et al.* (2003) [1+] used a split-plot design to evaluate management techniques for controlling *Molinia caerulea* and enhancing *Calluna vulgaris* in M15 wet heath in Northumberland. Burning was the core treatment of the study but two grazing treatments were also applied: heavy and light sheep grazing at 1.5 and 0.66 ewes/ha, respectively, which were prescribed for upland dwarf shrub heath under the former ESA scheme.

Calluna was replaced as the dominant species by *Molinia*, *Deschampsia flexuosa* and *Carex nigra* depending on the grazing rate: the grasses were favoured by light grazing and *Carex nigra* by heavier grazing. The results emphasise that initial vegetation composition influences changes in dominance among key species in wet heaths following burning and that management of the grazing regime may be the most effective means of suppressing ‘undesirable’ grasses and thereby favouring dwarf shrubs. Data from the study were included in the systematic review by Stewart *et al.* (2004) who interpreted the changes 3 years after burning as: a decline in *Calluna* and an increase in *Molinia* grazed at both stocking rates.

Marrs *et al.* (2004) [1+] conducted a six-year factorial RCT experiment to test a range of management treatments including grazing (year-round, summer and ungrazed) and burning (and herbicide) designed to reduce *Molinia* and encourage the development of dwarf-shrub vegetation at sites in the Peak District and Yorkshire Dales (described further under sub-question a).

Grazing tended to shift the community of ‘grey’ moors towards bog-moorland types but had little effect on ‘white’ moors. Grazing levels in the study were low and differences between summer only and year-round treatments were not seen in practice due to stocking patterns. It was suggested that at ‘white’ sites where *Molinia* is dominant a package of management measures including initial *Molinia* control, surface treatment to disturb the litter layer and addition of seed will be required to restore dwarf-shrub cover. It was suggested that a range of communities should be considered as end-points, especially where *Molinia* forms mixtures with dwarf shrubs as treatment may cause a shift towards acid grassland and that once mixed *Molinia*–*Calluna* vegetation is established, appropriate management by grazing and burning should be applied.

Reviews on vegetation change and burning and grazing interactions

Shaw *et al.* (1996) [4+] carried out a comprehensive literature review of the effects of burning and grazing of blanket bog and upland wet heath. Though grazing was reviewed in detail, little consideration seems to have been given specifically to the interaction between burning and grazing. They concluded that “the intensity and extent of damage through grazing varies according to stocking rates, wetness and condition of the site, type of grazing animal, time of year and length of time on the

site. The effects of grazing on the vegetation can vary depending on the other habitats and food supplies available to herbivores. There are therefore interactions between management history and grazing impact which can be difficult to separate."

Tucker (2003) [4+] in a review of burning in the uplands (described earlier under sub-question a) reviewed the interactions of burning and grazing and other management measures across upland habitats. He noted that "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 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. But these aspects of interactions ... have been little researched and are not understood in detail." He further suggested that "in general grazing and burning impacts appear to affect upland vegetation in similar ways. For example, increased burning frequencies and grazing intensities on heathland both tend to change vegetation from *Calluna*-dominated communities to grasslands. 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."

Anderson *et al.* (2006) [4+] assessed the potential role of burning as a management tool in the restoration of *Molinia*-dominated upland blanket mire in Wales and specifically the Elenydd Estate in mid-Wales, based on a literature review and fieldwork on the estate.

It was concluded that evidence from experimental studies demonstrates that burning without adequate grazing, or over-frequent burning with or without grazing, leads to what can become overwhelming dominance by *Molinia* over large tracts of land. This is a reflection of the ability of *Molinia* to exploit increased nutrient availability following a burn; its inherently high relative growth rate; its ability to sequester nutrients for rapid recycling early in the year; and its basal internodes which are protected from damage by fire. Control of *Molinia* in post-burn vegetation is most effective using less selective grazers such as cattle and ponies, particularly during summer.

Appendix 10 The studies and evidence on the relationship between burning and wildfire

No studies were found that specifically provided evidence on the direct relationship between managed burning and occurrence and severity of 'wildfire' in the UK. However, relevant aspects of two studies and two reviews related to wildfire risk and hazard are presented below (in date order).

Albertson *et al.* (2009, 2010) [2+]. Albertson *et al.* (2009) developed a 'probit' model to assess the chance of wildfires at different times of year, days of the week and under various weather conditions. Albertson *et al.* (2010) used the model to investigate the likely impact of climate change on the number of wildfires in the Peak District.

The Peak District is expected to experience warmer, wetter winters and hotter dry summers. Simulations of likely future weather applied to the model suggest an overall increase in occurrence of summer wildfires. Little change in wildfire incidence was predicted in the near future, but as climate change intensifies, the danger of summer wildfires is projected to increase from 2070. Albertson *et al.* (2010) suggested, therefore, that fire risk management will be necessary in future. In addition, that "moorlands may have to be managed to reduce the chance of summer wildfires becoming catastrophic ... and management measures may include controlled burning, grazing or mowing to remove fuel." These studies include, but do not relate specifically to upland peatlands, nor does the model consider the effect of habitat/vegetation type and structure on fire risk/hazard and severity. Albertson *et al.* (2010) do, however, consider the effect of land management on vegetation and mention on the one hand the potential of managed burns to reduce fuel loading and on the other, the other the potential of reduced burning coupled with restoration such as rewetting to improve peatland resilience to wildfire.

Reviews

Aylen *et al.* (2007) [4-] reviewed strategies for mitigating moorland wildfire risk in the Peak District. They sought to identify the best option in economic terms given the uncertainty surrounding outbreaks of moorland wildfires.

The authors noted that "at one extreme, 'zero tolerance' is considered an expensive but efficient way to tackle fires that do occur." They suggested that in practice there is some optimum level of fires presumably determined by some cost-benefit exercise. Amongst possible measures suggested to reduce risk of fires breaking out, controlled burning in winter was listed along with controlling access, education, fire watching, fire breaks, preventative wetting of dry vegetation, and grip blocking. The authors recommended a combination of fire fighting resources, with more emphasis on helicopter use, rapid deployment to more remote locations and risk planning. Also that attention should be paid to "planning for catastrophe as well as regular fire events".

Davies *et al.* (2008) [4+] reviewed the future of fire management in the British uplands including a case study on peatland soils. They identified a range of factors controlling fire hazard, risk and behaviour: fuel structure, weather, climate, physical environment, for example, altitude, topography, aspect and soil type, and human interference. On the last, they noted that "the fire regime in the UK is largely governed by humans. Historical use of fire means that flammable landscapes exist even in the absence of contemporary traditional burning; the UK is in a bioclimatic zone that, despite being oceanic, still allows for some 'natural' fires. The vast majority of fires are caused by people, however, and although we have some records of naturally occurring wildfires … such outbreaks are rare. Factors such as ease of access, holiday periods and proximity to urban areas may all be important in governing the risk of accidental fires." "Demand for traditional rural sporting pursuits, the status and

economics of agriculture, conservation and land management targets may also affect the amount of management burning and impact on vegetation, and thus the distribution and nature of fuels." Amongst the factors they list as driving changes in fire regime were: reduced grazing, increased risk of accidental ignitions from public access following the introduction of open access land as well as an increasing emphasis on management for, and promotion of, recreation; rising labour costs reducing resources for wildfire prevention and fighting, as well as for management and prescribed burning; and climate change scenarios including an increased frequency and intensity of periods of drought and "changing seasonal trends in fuel moisture content which could be extrapolated to mean an increase in fire hazard, although the issues are complex. It is important to remember, however, that fires will happen regardless of our intentions, and there is thus a need for us to think about manipulating fire regimes to achieve a range of management outcomes." Amongst the recommendations for future management are careful consideration of 'fuel management' and fire hazard. They argue that the idea of fuel management is now well established elsewhere in the world "and suggests that a network of prescribed burns to break up large areas of flammable vegetation are desirable from a fire control point of view, even when the management aim is to avoid fire where possible." These might include using fire to create fire breaks and control zones "that can prevent whole landscapes being lost in a single wildfire", when considering fire effects on the carbon cycle, removing the arbitrary and variable peat depth distinction and including all soils with an organic carbon content greater than 14% (25% organic matter).

McMorrow *et al.* 2009 [2+] reviewed moorland wildfire risk, visitors and climate change using the Peak District National Park (PDNP) as a detailed case study. This included detailed descriptions of modelling work carried out on the effects of climate change, timing of wildfires and the spatial distribution of wildfires. The last used an empirically informed and stakeholder-guided modelling methodology to estimate the likelihood of wildfire occurrence based on relationships between spatially explicit wildfire indicators and previously reported wildfire events. Habitat type was used as an indicator of the likelihood of fire occurrence by approximating the potential vegetative fuel loading and the type and wetness of the substrate. Three classes emerged as being associated with a higher than predicted number of wildfires: bare peat, eroding moor and inorganic bare ground. Although large numbers of wildfires occurred on heather moorland this reflects their large extent and occurrence was lower than expected. It was suggested that "a reason for the relatively low empirical risk on heather moor may be that managed fires reduce subsequent fuel load for wildfires. The database also did not allow fire intensity or fire hazard to be modelled, only the risk of fire occurrence, whereas stakeholders may have been influenced by the higher fire hazard presented by fine fuels like heather." "Conclusions regarding land management are controversial and echo the fire paradox approach now common in the USA and in southern Europe (for example, McKenzie *et al.* 2004). Spatial analysis for the PDNP shows that heather moorlands are less prone to reported wildfire outbreaks (McMorrow *et al.* 2006, McMorrow & Lindley 2007)." The reason for this is not clear, though it was suggested that "it may be that small escaped fires on heather are dealt with by land managers and so are not reported, but large escaped fires are unlikely to remain unnoticed. Burning to reduce the fuel load could therefore be seen as an effective, if controversial, management solution on heather moors (burning on UK blanket bog is normally not permitted)." However, the model does not consider the actual relationship with managed burning per se, nor apparently, the distinction between *Calluna*-dominated 'heather moorland' on deep peat or mineral soil.

Appendix 11 The studies and evidence on the extent, frequency and practice of managed burning

Nine studies all from the England provide evidence on sub-question (h). Most are national sample or census surveys, though others cover individual moorland areas: the Peak District NP/North Peak ESA, the North York Moors NP and the North Pennines AONB.

Remote sensing studies

ADAS (1997)/MAFF (1993) [2++] mapped the extent of moorland burning in the North Peak ESA between 1988 and 1995 based on aerial photographic interpretation (API). Mapping of the 'core ESA' was carried out by MAFF (1993) between 1988 and 1991 which was extended by ADAS (1997) between 1991 and 1995 within both the 'core' ESA and the 1993 extension areas. The accuracy assessment of the 1991–1995 API (ground-truthed for 230 burns at eight sites) revealed an overall mapping accuracy of 99%. Habitat maps were produced in 1988 and 1993 from a combination of API and ground checking. Overlaying maps of dry heath and 'dry bog' habitat and burning allowed examination of the pattern of burning in relation to habitat.

Within the original core ESA there was an increase in the number and area of burns and the proportion of 'heather moorland' burned annually (from 443 burns covering 179 ha in 1988/89 to 1,690 covering 490 ha between 1991 and 1995). Whilst the increase occurred on both ESA agreement land and non-agreement land, it was greatest on the former. Burning on the two heather-dominated habitat types, dry heath and 'dry bog' accounted for 93% of the total area of moorland burned from 1991 to 1995. Overall, similar proportions of dry bog (4%) and dry heath (3%) were burned annually (representing average rotations of 25 and 30 years including unburnt and unburnable areas).

Anderson *et al.* (2009) [2-] developed a series of models to test hypotheses about the factors influencing the distribution of a species of conservation importance, the hen harrier. As part of this, they produced a 'burn intensity index' (as a measure of gamekeeper activity) and map based on the proportion of heather burnt within 10 km grid squares based on API mostly of 2005-2006 aerial photographic images.

This indicated that in England, more intensive 'strip burning' of heather (on heath and bog) was largely restricted to the Pennines, Bowland, North York Moors and Northumberland, probably mainly on grouse moors.

Penny Anderson Associates (2012)/Natural England (2010) [2++] mapped the level of intervention/management on all upland, deep peat in England within the Defra Moorland Line (ADAS 1993) using API of 2012 (and some 2005) photos. The upland deep peat map was derived by Natural England from a variety of sources, primarily soil mapping (Soil Association mapping where the dominant series have >40 cm of peaty surface material and British Geological Survey (BGS) drift mapping indicated peat deposits) and where the Natural England BAP Priority Habitat Inventory indicated blanket bog habitat. Forestry and intensive agriculture were excluded. Various interventions/impacts were mapped including burning, gripping, hagging and erosion.

The mapping showed that 24% (76,991 ha) of the total extent of moorland deep peat in England was subject to burning. Of this, 22,063 ha (29%) was also gripped and an additional 3,161 ha (4%) also 'hagged', with a further 507 ha (0.7%) both gripped and hagged. There was significant regional variation, although no summary statistics were given on this other than descriptions by moorland

areas which indicate that most occurred in the north of England. Natural England (2010) mapped the extent of burning on peat soils in the North Pennines and Yorkshire Dales based on these data.

Yallop *et al.* (2005, 2006b) [2+] used API to map the location, extent and parcel size of burning in a 2% sample of 1-km squares (208) within the Countryside Survey 2000 (CS2000) Upland Environmental Zone in England from photos taken in 2000. They used a random systematic sampling design to give even coverage (with 2 random sample squares per 10-km square). Moorland was divided into just two recognisable types: 'dwarf shrub heath' and 'graminoid-dominated' (both including areas on deep peat). Blanket bog and other peatlands were not separated from these two types. Burns were classified in to four recognisable age ranges and the burn 'return period' was estimated from burn age and the proportion burnt. Change between the 1970s and 2000 was assessed in a subsample of squares (23) within upland National Parks.

Visible evidence of burning on APs was only consistently found in *Calluna*-dominated stands (including but not separating those on blanket peat). These occurred in 51% (106) of sample squares, though only covered 24% of the total area. Burning occurred in 71% (75) of these stands with an average of 27% showing visible evidence of burning. When only squares showing evidence of recent burning were considered the average area in burning management increased to 38%. Median burn parcel size was 0.25-0.28 ha for parcels in the new and recent burn classes. The median estimated return period for 'consistently managed sites' was 28 years with the most frequent range class 16-20 years, though there was considerable proportion above and below this. There was a significant increase in the extent of new burns within the NP subsample between the 1970s and 2000 (from 15.1% to 29.7%, $p < 0.001$) with a corresponding significant decline in the mid-aged burn class (25.7% to 15.8%, $p < 0.05$). Though not separating out blanket bog, the sample survey provided the first national estimate of the extent, size and return period of burning on 'heather moorland' in England.

Yallop *et al.* (2006a) [2+] mapped the extent of burning on blanket bog and upland heathland in the North Pennines AONB on a 35 m sampling grid from API of photos taken in 2001-03. Four ages classes of burn were again identified but, due to poor quality of the photography, these were amalgamated to form just two classes: 'new burn' and 'no visible burn'. Ground truthing was not considered possible given that the photos were up to five year old, but consistency of independent interpretation of the two amalgamated classes in 1,000 random sample points was 96%.

The results indicated that 20.7% of the total area of blanket bog and upland heathland in the Natural England Priority Habitat Inventories in the North Pennines AONB was 'recently' burnt, comprising 19.2% of blanket bog and 27.5% of upland heathland (the latter probably including at least a proportion of wet heath). About half of the area of bog and c.28% of the heath showed no evidence of burning. There was considerable variation in estimated repeat time, but c.20% of the bog was burnt with a repeat time of under 20 year. At least 30% of the bog in upland SSSIs in the AONB was under intensive burning regimes with repeat times of less than 20 years.

Yallop *et al.* (2012) [2++] remapped the extent of burning on blanket bog and deep peat soils nationally (using 2000 aerial photos) and in the North Pennines AONB (2001-03) based on re-examination and reconciliation of data generated in previous studies by Yallop *et al.* (2005, 2006a, b; see above) and unpublished data held by Natural England from mapping of new moorland burning in the entirety of the Peak District NP in 2005 and the North York Moors NP in 2009 using image segmentation techniques on 4-band digital aerial imagery. Due to differences in the reported duration of burn age classes, the extent of burning was summarised as annualised rates to allow comparisons between the different areas. Burn mapping from these studies was overlain on a peat soil dataset (National Soil Resource Institute deep and blanket peat soil series), for the first time, and the English Nature Blanket Bog (and Upland Heathland) Priority Habitat Maps, for the first time for the national dataset.

The results indicated that the percentage of the total extent of blanket bog habitat (upland heathland habitat for comparison in brackets) burnt annually was 1.5% (3.0%) in the national sample dataset. Corresponding figures for the other areas were: 4.7% (5.4%) in the North York Moors, 2.5% (3.6%) in

the North Pennines and 1.4% (1.9%) in the Peak District. If only the heather-dominated areas within the habitats (which are the most 'burnable' areas of the habitats and the parts where burning is most easily detectable from API) are considered, the figures are higher: 3.8% (4.5%) in the national dataset, 8.5% (7.5%) in the North York Moors, 6.7% (6.2%) in the North Pennines and 4.0% (4.2%) in the Peak District. The figures are similar for deep peat soils, eg1.5% of deep peat nationally (3.7% of the heather-dominated area on deep peat).

These figures give burn 'repeat times' for blanket bog habitat (heather-dominated part in brackets) of: 64 year (26.5 year) in the national dataset, 21.2 year (11.7 year) in the North York Moors, 39.3 year (15 year) in the North Pennines and 73.1 year (25 year) in the Peak District.

The rate of burning on SSSIs (1.8% of blanket bog habitat and 4.0% of the heather-dominated area) was slightly greater than on non-SSSI (1.1% and 3.3%) in the national dataset. In the national sample, 38% of all burning was on blanket bog habitat and 46% on deep peat.

Habitat condition surveys

Critchley (2011a) [2++] surveyed the condition of blanket bog priority habitat in England in 2008 and 2009 based on a stratified random sample of 97 habitat polygons from the Natural England Blanket Bog Priority Habitat Inventory using the upland Common Standards Monitoring (CSM) methodology (JNCC 2009). Polygons were stratified by designated site status (SSSI and non-SSSI) and agri-environment (AE) agreement status (agreement and non-agreement) and randomly sampled with 37 points. The field assessments included two CSM attribute targets directly related to burning: "no burning into the bryophyte and lichen layer" and no burning in 'sensitive areas' (as defined in the CSM guidance).

None of the 97 sample sites were in favourable condition. The burning into the bryophyte and lichen layer attribute target ('Target 10') was not met in 21% of sites and the burning in sensitive areas attribute ('Target 11') in 15% (based on 'blanket bog' samples on peat >30 cm and the CSM threshold for a site/feature passing being 90% of samples passing all the targets). Overall, 11% of the 2,196 blanket bog samples across sites did not meet the burning into the bryophyte and lichen layer target. Pass rates were significantly lower for SSSI than non-SSSI sites (59% and 96% for T10 and 66% and 98% for T11, both $p < 0.001$), though slightly higher for AE agreements than non-agreements (81% and 77% for T10 and 88% and 79% for T11, both ns). The pass rate was higher for SSSI sites under AE agreements than not ($p < 0.001$). Although higher than the pass rates for some of the other attributes recorded (for example, no sites passed the frequency of indicator species target), they are relatively low given that when peatlands are burnt, it would be expected that only a proportion would be burnt each year (for example, 6.7% based on a 15-year rotation), that the sample includes unburnt sites and that the 'cool burns' generally advocated should be to avoid burning into the bryophyte and lichen layer. However, at least some of the burns may have related to wildfires. These data suggest that the severity of burns is variable, both within and between sites and that in practice burns into the bryophyte and lichen layer occurs in a proportion of cases.

Critchley (2011b) [2++] similarly surveyed the condition of wet heath as part of upland heathland priority habitat in England in 2009 and 2010 based on a stratified random sample of 99 habitat polygons from the Natural England Upland Heathland Priority Habitat Inventory using the same stratification and CSM methodology. Peat depth was used to classify sample points as wet (>20 cm) or dry heath (≤20 cm) and the corresponding CSM attribute targets applied to each point. The field assessments for wet heath included the same two CSM attribute targets directly related to burning as for blanket bog.

None of the 99 heathland sample sites were in favourable condition overall (wet and dry heath combined), nor the 88 with separate wet heath assessments. The burning into the bryophyte and lichen layer attribute was not met in 31% of sites and the burning in sensitive areas attribute in 12% (based on the CSM threshold for a site/feature passing being 90% of samples passing all the targets). Overall, 17% of the 927 wet heath samples across sites did not meet the burning into bryophyte and lichen layer target. The SSSI and AE agreement status of sites had no significant

effect on the pass rate for the burning in to the bryophyte and lichen layer target, but the pass rate for SSSI sites was significantly higher than for non-designated sites and there was a significant interaction with AE agreements (both $p<0.05$).

Other data on burning activity

Natural England (2009) [2+] produced an uplands atlas presenting on GIS and other geographical data and information on the environment and associated ecosystem services in English uplands. This included information provided by the Moorland Association on changes in grouse shooting activity over an eight-year period between 2001 and 2009.

This showed increases of 28.5% in the number of gamekeepers employed (200 in 2001 and 257 in 2009) and 29.4% in the potential number of shooting days per year (1,492 and 1,930) across the north of England. There were differences between areas with the increases greatest in the increases in the number of gamekeepers mainly in the Peak District, North Pennines, Yorkshire Dales and North York Moors.

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